Miscellaneous Paper EL-94-6
June 1994

AD-A283 087

(1)

US Army Corps
of Engineers
Waterways Experiment
Station

Water Quality Studies: Richard B. Russell and J. Strom Thurmond Lakes; Summary Report

*by Steven L. Ashby, Robert H. Kennedy,
Joe H. Carroll, John J. Hains*

WES

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Prepared for U.S. Army Engineer District, Savannah

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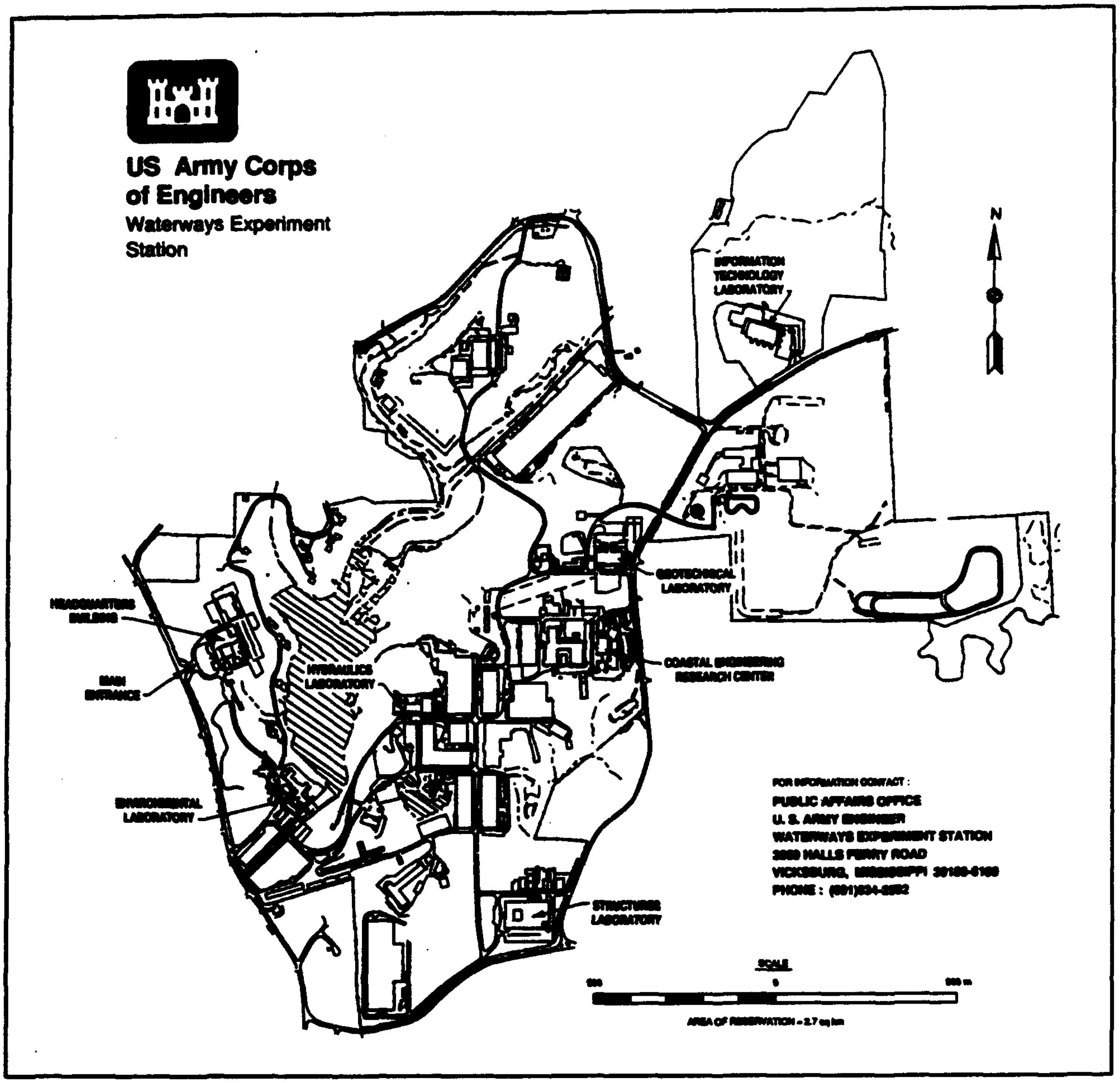
Final report

Prepared for U.S. Army Engineer District, Savannah
Savannah, GA 31402-0889



Waterways Experiment Station Cataloging-in-Publication Data

Water quality studies : Richard B. Russell and J. Strom Thurmond Lakes; summary report / by Steven L. Ashby ... [et al.] ; prepared for U.S. Army Engineer District, Savannah.

82 p. : ill. ; 28 cm. — (Miscellaneous paper ; EL-94-6)

Includes bibliographic references.

1. Water quality — Richard B. Russell Lake (Ga. and S.C.) — Testing. 2. Richard B. Russell Lake (Ga. and S.C.) 3. Water quality — J. Strom Thurmond Lake (Ga. and S.C.) — Testing. 4. J. Strom Thurmond Lake (Ga. and S.C.) I. Ashby, Steven L. II. United States. Army. Corps of Engineers. Savannah District. III. U.S. Army Engineer Waterways Experiment Station. IV. Environmental Laboratory (U.S. Army Engineer Waterways Experiment Station) V. Series: Miscellaneous paper (U.S. Army Engineer Waterways Experiment Station) ; EL-94-6.
TA7 W34m no.EL-94-6

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Preface

A multiphase, comprehensive water quality study at Richard B. Russell, J. Strom Thurmond, and Hartwell Lakes was initiated in October 1983 as a cooperative effort by the U.S. Army Engineer District, Savannah (SAS), and the U.S. Army Engineer Waterways Experiment Station (WES). This report, which covers the period January to December 1988, is the fifth in a series of annual interim reports documenting findings and results. It is a summary report and is submitted in accordance with the "Scope of Work: Water Quality Monitoring Program - Richard B. Russell Dam and Lake, Georgia and South Carolina" (Intra-Army Order No. PD-EI-84-07).

This report was prepared by Mr. Steven L. Ashby, Dr. Robert H. Kennedy, Mr. Joe H. Carroll, and Dr. John J. Hains of the Environmental Laboratory (EL), WES. Dr. Kennedy and Mr. Carroll were responsible for the conduct of the study and for preparation of this report. The work was conducted under the direct supervision of Dr. Richard E. Price, Acting Chief, Ecosystem Processes and Effects Branch, and under the general supervision of Mr. Donald L. Robey, Chief, Environmental Processes and Effects Division, and Dr. John W. Keeley, Director, EL.

Dr. Robert W. Whalin was Director of WES during the preparation of the report. COL Bruce K. Howard, EN, was Commander.

This report should be cited as follows:

> Ashby, S. L., Kennedy, R. H., Carroll, J. H., and Hains, J. J. (1994). "Water quality studies: Richard B. Russell and J. Strom Thurmond lakes; Summary report," Miscellaneous Paper EL-94-6, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Conversion Factors, Non-SI to SI Units of Measurement

Non-SI units of measurement used in this report can be converted to SI units as follows:

Multiply	By	To Obtain
miles (U.S. statute)	1.609347	kilometers
tons (short, 2,000 lb)	907.1847	kilograms

1 Introduction

Water Quality Management Concerns

The multi-use nature of reservoirs often necessitates diverse management to maintain maximum potential benefits of these national resources. Historically, reservoirs have been managed and operated primarily to provide flood control, hydroelectric power generation, and water supply. However, in recent years, utilization of reservoirs has been expanded to include fish and wildlife habitat, water quality control, and water-based recreation. Increasing emphasis on the environmental and recreational value of reservoirs has resulted in increased public demand for the protection of such a resource. This, in turn, has prompted efforts to better understand reservoir water quality processes and to develop sound management strategies for the protection of this resource.

Limnologists and others concerned with the management of reservoir water quality evaluate water quality processes relative to influences of reservoir operations and resultant impacts on water quality. Commonly encountered water quality management concerns are hypolimnetic anoxia, the accumulation in bottom waters of reduced metals such as iron and manganese, and excessive rates of algal production. In cases in which operational procedures require the withdrawal of water from anoxic bottom strata, downstream areas are exposed to elevated metal, nutrient, and hydrogen sulfide levels and the influx of oxygen-deficient release waters. Such conditions impact aesthetic, recreational, and water-use values and threaten fish and wildlife habitat.

Objectives of Long-Term Study

The construction of Richard B. Russell Dam and the impoundment of the 48-km reach of the Savannah River between Hartwell Dam and J. Strom Thurmond Lake (formerly called Clarks Hill Lake) raised concerns over water quality conditions in the newly filled Richard B. Russell Lake and the potential impact of reservoir releases on the water quality of J. Strom Thurmond Lake. Accordingly, these concerns were addressed as major objectives of an intensive water quality study to (a) describe post-impoundment water quality conditions in Richard B. Russell Lake; (b) document the impacts of the new impoundment on water quality conditions in J. Strom Thurmond Lake, and

(c) evaluate the effectiveness of the oxygenation system in ameliorating potential water quality problems in Richard B. Russell Lake and its tailwater.

Scope

The U.S. Army Engineer Waterways Experiment Station, through cooperative agreement with the Savannah District of the South Atlantic Division of the Corps of Engineers, began this investigation in October 1983. This report documents the results of the fifth year of the investigation and summarizes the major results of 5 years of observation regarding the above-mentioned objectives. The study involved a combination of routine and event- or process-oriented data collection efforts in Richard B. Russell and J. Strom Thurmond Lakes at representative sampling stations established in the lakes and their tailwaters. The forebay region and tailwater of Hartwell Lake were also sampled on a routine basis. Other studies of specific events or processes influencing water quality were conducted over shorter time periods and with greater sampling intensity.

Also, a special study to identify sediment deposition in selected regions of Richard B. Russell Lake and J. Strom Thurmond Lake was conducted in 1988.

2 Site Description

The Savannah River Basin is long and relatively narrow with its long axis lying in a northwest-southeast direction. The maximum length of the basin is nearly 402 km, while the maximum width is approximately 113 km. The total area of the basin is 27,400 km^2. The Savannah River originates on the southern slope of the Blue Ridge Mountains in North Carolina, and flows in a southeasterly direction through the Piedmont Plateau and Coastal Plain along the boundary between Georgia and North and South Carolina.

Prompted by the need for flood control, streamflow regulation, and water supply in the basin, the U.S. Army Corps of Engineers constructed Clarks Hill Dam (now known as J. Strom Thurmond Dam and Reservoir) in 1954. Hartwell Dam, completed in 1963, extended impoundment of the Savannah River into its two major tributaries, the Tugaloo and Seneca Rivers. Richard B. Russell Dam and Reservoir was authorized as Trotters Shoals Dam on 7 November 1966 by the "Flood Control Act of 1966" (Public Law 89-789, Eighty-Ninth Congress HR 18233) to provide power generation, incidental flood control, recreation, fish and wildlife habitat, streamflow regulation, and water supply. Richard B. Russell Dam, completed December 1983, impounds the Savannah River between Hartwell and J. Strom Thurmond Lakes (Figure 1). Major physical and morphometric features of the three impoundments are listed in Table 1. Descriptions of the basin and associated landforms are presented in Design Memorandum 8 for Russell Dam and Reservoir (U.S. Army Engineer District, Savannah 1974) and summarized below.

Topography in the basin is characterized by gently sloping upland areas cut by gullies and stream valleys. Relief is more rugged in areas adjacent to the Savannah River where well-developed, moderately steep to steep ridges and ravines form the topography. Stream elevation varies from 274 m, referred to the National Geodetic Vertical Datum (NGVD), above Hartwell Dam to 55 m NGVD below J. Strom Thurmond Dam.

Rock formations of the mountain section and the Piedmont Plateau are primarily igneous or metamorphic in origin, and include granites, gneisses, schists, basic eruptives, and highly metamorphosed shales, sandstones, and limestones. On most level or gently sloping areas, the rocks are disintegrated

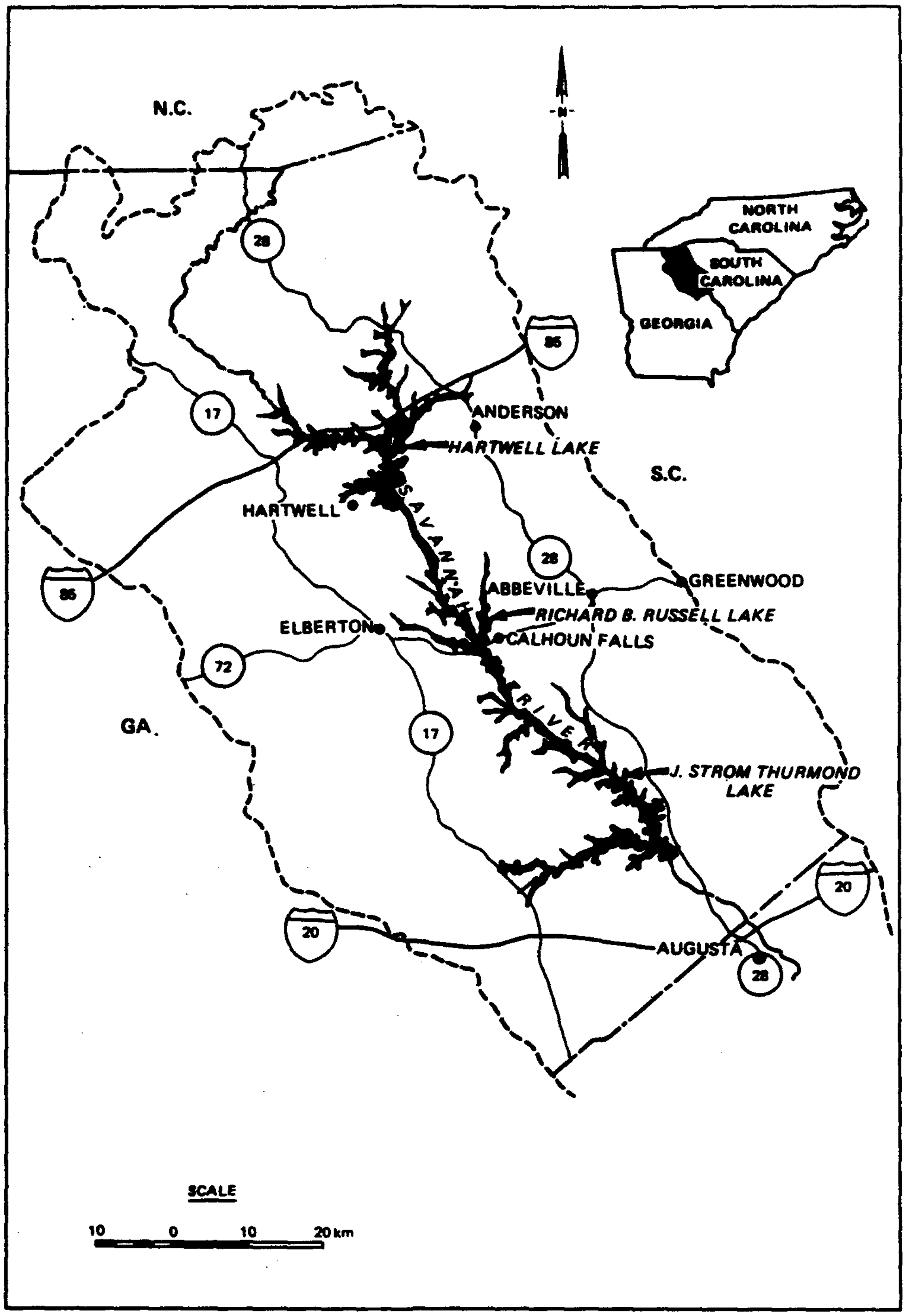

Figure 1. Locations of Hartwell, Richard B. Russell, and J. Strom Thurmond Lakes in the Upper Savannah River watershed

to a depth of many meters and the surface is largely formed of residual material. Exposed rock outcrops are mostly confined to summits, steep slopes, and stream channels.

Table 1
Physical Characteristics of J. Strom Thurmond, Richard B. Russell, and Hartwell Lakes

Characteristics	J. Strom Thurmond Lake	Richard B. Russell Lake	Hartwell Lake
Max power pool el, m NGVD	100.6	144.8	201.2
Min power pool el, m NGVD	95.1	143.2	190.5
Flood control pool el, m NGVD	102.1	146.3	202.7
Surface area,[a] km^2	283.2	107.9	226.6
Volume,[a] 10^6 m^3	3,023.3	1,271.0	3,146.7
Maximum depth,[a] m	45	47	55
Mean depth,[a] m	11	12	14
Reservoir length,[a] km	63	45	79 Tugaloo R. 72 Seneca R.
Shoreline length,[a] km	1,930	885	1,548
Shoreline development ratio[a]	32.0	24.5	29.0
Drainage area,[a] km^2	14,906	7,508	5,406
Residence time,[a] day	144	102	306
Mean discharge, m^3 s^{-1}	243.6[b]	143.8[c]	119.0[b]
Top of penstock el, m NGVD	88.4	133.2	181.4
Bottom of penstock el, m NGVD	69.5	110	160.9

[a] At maximum power pool elevation.
[b] Mean discharge for the period 1925-1973.
[c] Mean river discharge for 31 years.

The basin experiences a maritime climate with mild winters, long summers, and moderate to high rainfall (114 to 152 cm per year). While subject to continental influences, it is protected by the Blue Ridge Mountains from the more rigorous winters prevailing in the Tennessee Valley. The mean temperature for the basin is approximately 16 °C. During the coldest months of the year, night temperatures frequently reach approximately -7 °C. During the warmest months of the year temperatures often exceed 32 °C. At lower elevations, the winters are milder and the summer temperatures greater.

Land use is fairly uniform throughout the basin with woodlands comprising 60 percent, and pasture and cropland about 35 percent of the area. A small (5 percent), but growing, portion of the watershed is exposed to urban and recreational use. The wooded uplands consist of mixed pine and hardwood forest, whose timber provides one of the major industries of the area. Cleared lands are devoted primarily to cultivated crops and pasture.

The waters of the Savannah River above J. Strom Thurmond Dam are relatively free of pollution. Current point sources of pollution are minimal with most occurring in the Hartwell Lake area. Contamination with polychlorinated biphenyls (PCBs) in Hartwell Lake has been reported (Gaymon 1982). Numerous small wastewater treatment plants, textile mills, and various other industries discharge to streams and lakes within the study area. Due to the abundance of granitic rock, poorly drained soils, and the steepness of slopes, surface waters tend to have low dissolved solids, low alkalinity, and a low buffering capacity.

3 Sampling and Analytical Methods

Detailed descriptions of the sampling methods and analytical porcedures used during the fifth year of the study are presented in previous annual reports (James et al. 1985, 1986). The locations of primary sampling stations in Hartwell and Richard B. Russell Lakes and in J. Strom Thurmond Lake are presented in Figures 2 and 3, respectively. In situ data were collected monthly to describe temporal and spatial patterns of temperature, dissolved oxygen concentration, and specific conductivity. Water samples for physicochemical and biological analyses were collected seasonally. Four seasonal sampling times coincided with the spring high flow period, the early and late stratification periods, and the period following fall mixing. Table 2 lists the types of routine analyses made during the study.

In addition to routine water quality monitoring, the area near Richard B. Russell Dam was intensively sampled to describe influences of the oxygenation system. From the onset of thermal stratification until November, weekly in situ measurements were collected at sta 060B, 100B, 115, and 120. Monthly physicochemical data were also collected at sta 040, 045, 050, 060B, 100B, and 120. Additional stations and dates were often included to supplement these data. Temperature, dissolved oxygen, pH, specific conductance, and oxidation-reduction potential data were collected in situ using a Hydrolab Surveyor II (Hydrolab Corp., Austin, TX). Monitoring instruments were calibrated prior to each field use. Temperature was calibrated to the nearest 0.1 °C against a National Bureau of Standards (NBS) certified thermometer; dissolved oxygen by air calibration; pH with standard buffer solutions; and specific conductance using known standards. Because it was interpreted primarily as an indicator of reducing versus oxidizing chemical conditions, and not a quantitative measure, minimal emphasis was given to calibration for oxidation-reduction potential. In situ measurements were also collected and recorded hourly at sta 010, 050, and 200 with Schneider Water Quality Monitors (Model RM25, Schneider Instrument Company, Madeira, Cincinnati, OH).

Water samples for physicochemical analyses were collected using a hose and pump at selected depths throughout the water column at each lake station. Samples for biological analyses were collected as integrated samples from surface to a depth equal to twice the Secchi disk depth. Sampling and

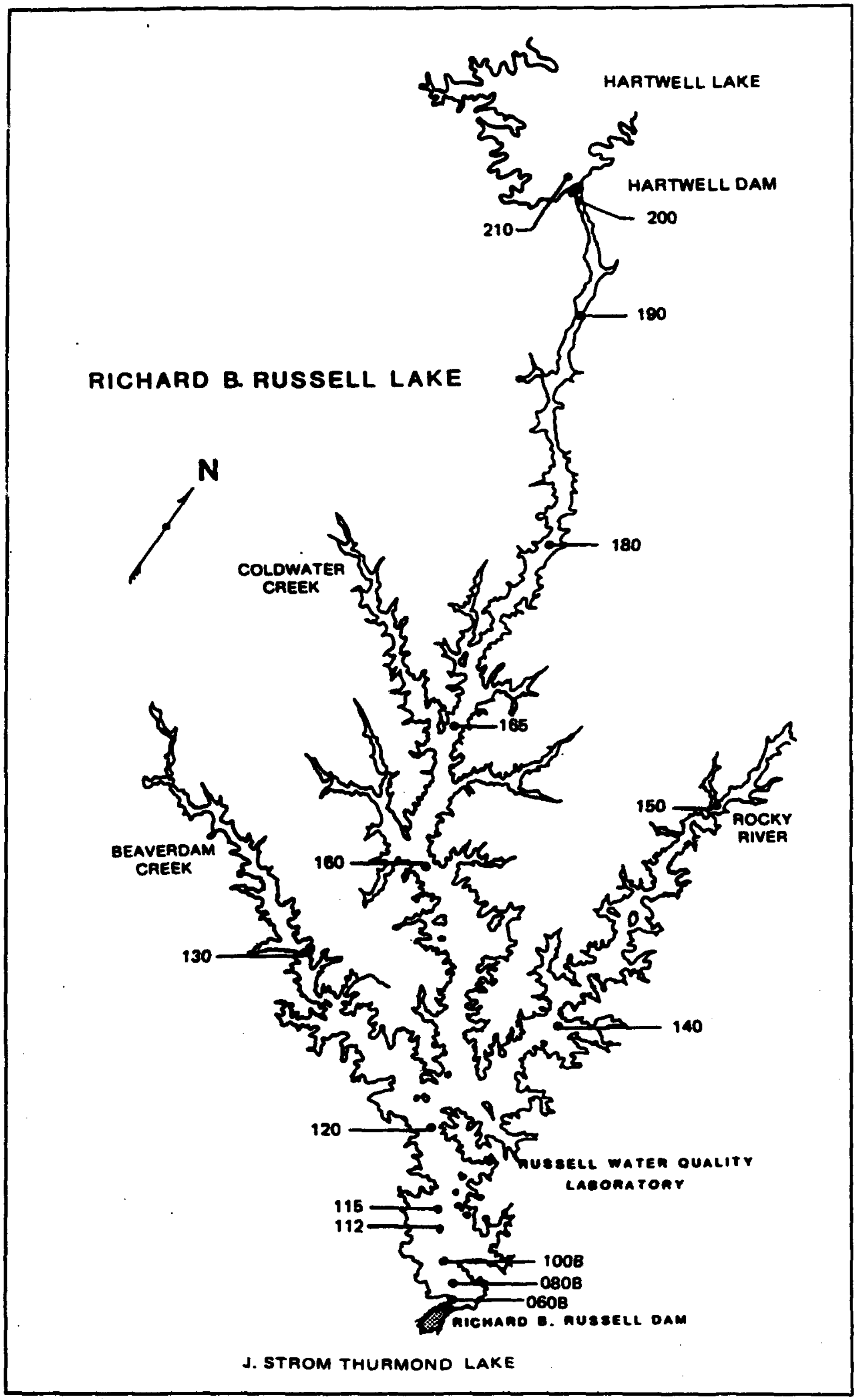

Figure 2. Locations of sampling stations on Hartwell and Richard B. Russell Lakes

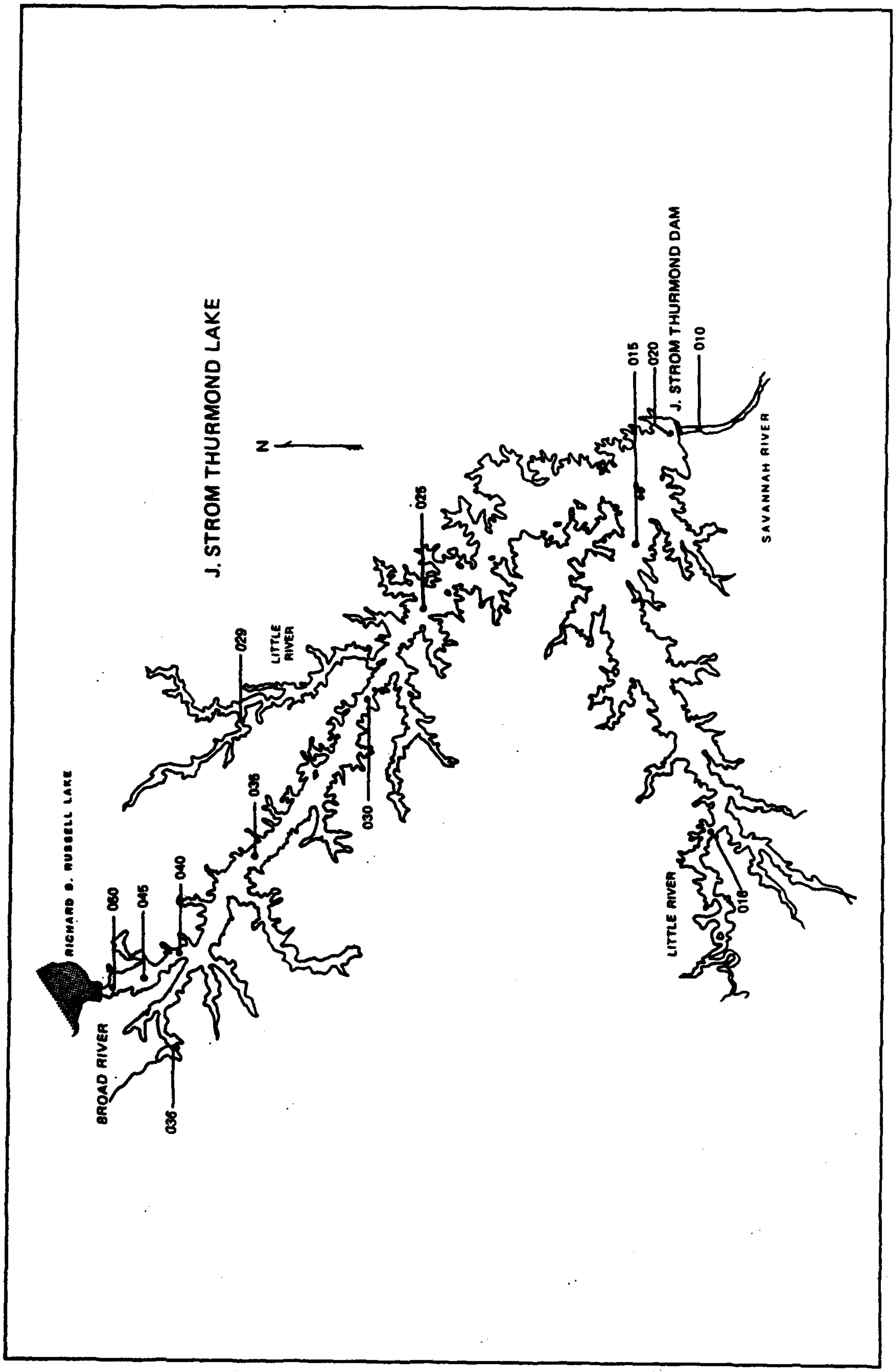

Figure 3. Locations of sampling stations on J. Strom Thurmond Lake

Table 2
Variables for Which Samples Were Analyzed

In Situ	**Nutrients**
Temperature (T)	Total Organic Carbon (TOC)
Dissolved Oxygen (DO)	Dissolved Organic Carbon (DOC)
pH	Total Phosphorus (TP)
Specific Conductance (COND)	Total Soluble Phosphorus (TSP)
Oxidation-Reduction Potential (ORP)	Soluble Reactive Phosphorus (SRP)
Secchi Disk Transparency	Total Nitrogen (TN)
Physicochemical	Total Dissolved Nitrogen (DN)
Total Alkalinity	Ammonia Nitrogen (NH4-N)
Sulfide	Nitrate/Nitrite-Nitrogen (NO3/NO2-N)
Metals	
Total Iron (TFE)	
Dissolved Iron (DFE)	
Total Manganese (TMN)	
Dissolved Manganese (DMN)	
Biological	
Chlorophyll *a*	

analytical methods, digestion and filtration techniques, sample holding times, and quality control statistics are presented in Appendix A. Standard methods (American Public Health Association 1980; U.S. Environmental Protection Agency 1979) were used for laboratory analyses of water samples.

4 Results of Study

Hartwell Lake

Pool elevation, precipitation, mean daily inflow, and mean daily discharge (outflow) for Hartwell Lake during 1984-1988 are depicted in Figure 4. Hydrologic conditions in 1988 were similar to those of 1986 and 1987, reflecting seasonal drought conditions. While pool elevations returned to near-normal levels during the 1987-1988 winter period, levels during the summer decreased to 1986 and 1987 summer levels due to below-normal inflows and operational requirements for release.

Monthly temperature data collected during 1988 at the Hartwell Lake forebay (sta 210) depict seasonal changes in the development of vertical thermal structure in the lake (Figure 5). Isothermal conditions, with temperatures ranging from 8 to 10 °C, were observed from January until mid-April. Surface warming and the onset of thermal stratification were observed by late April and early May, with establishment of a well-developed thermocline, at a depth near 12 m, occurring by early May. Surface temperatures continued to increase to near 28 °C throughout the summer season; however, the thermocline remained near a depth of 10 to 14 m. Hypolimnetic temperatures during the period of stratification ranged from 10 to 16 °C. Seasonal cooling in late September and early October resulted in a gradual weakening of thermal structure and, by mid-November, fall mixing had returned the lake to near isothermal conditions with temperatures between 12 and 14 °C.

Temporal and vertical trends in dissolved oxygen concentrations were observed at sta 210, coincident with the development of thermal gradients (Figure 6). While remaining near 8 to 10 mg/ℓ during isothermal conditions, dissolved oxygen concentrations began to decline in the hypolimnion as temperature gradients were established. Hypolimnetic dissolved oxygen concentrations decreased from near 6.0 mg/ℓ in early July to near 2.0 mg/ℓ in September. Anoxic conditions became established in bottom waters in October and persisted until late-December. However, following fall mixing in mid-November, dissolved oxygen concentrations were near 8 mg/ℓ throughout most of the water column.

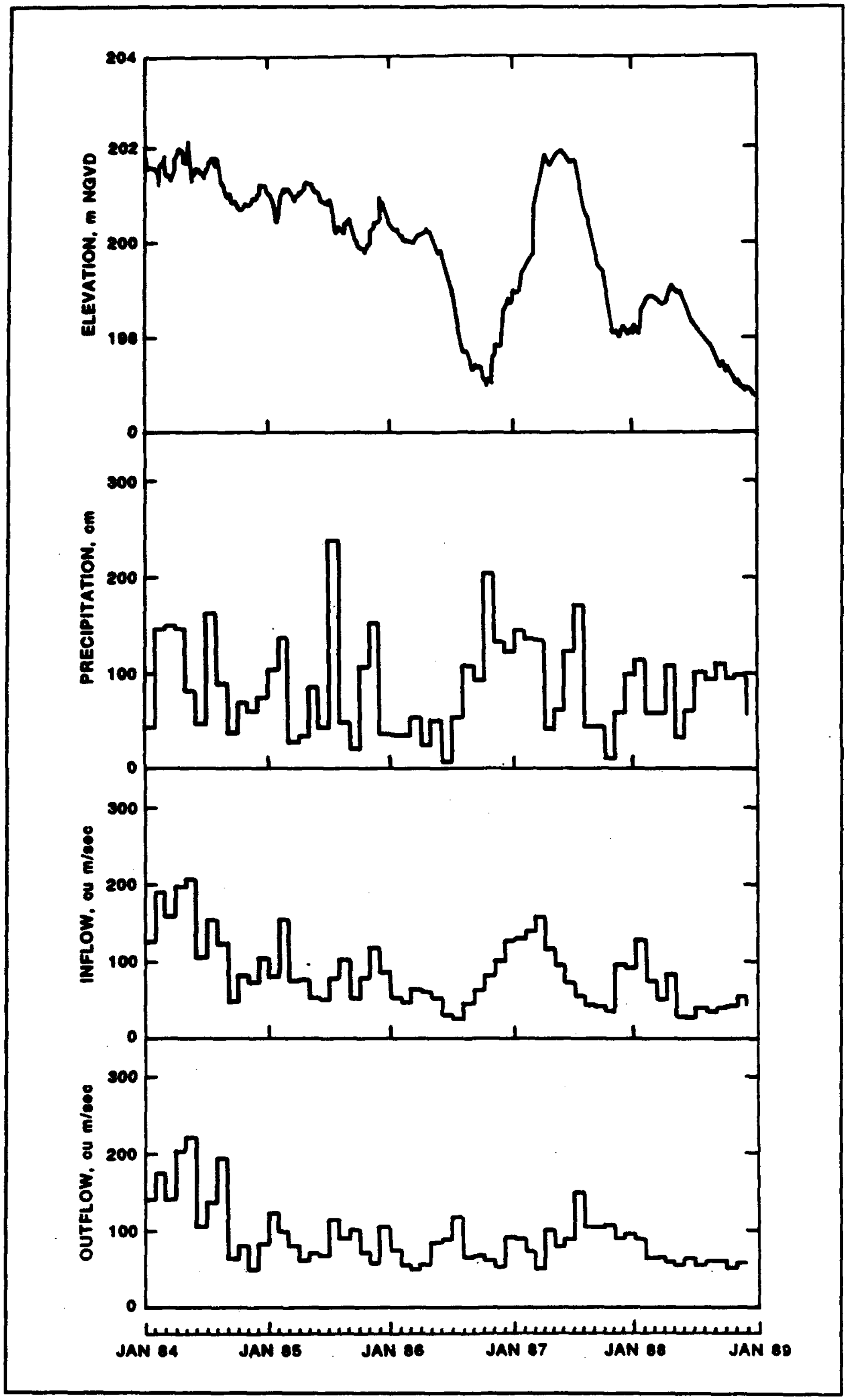

Figure 4. Pool elevation, precipitation, inflows, and outflows for Hartwell Lake, 1984-1988

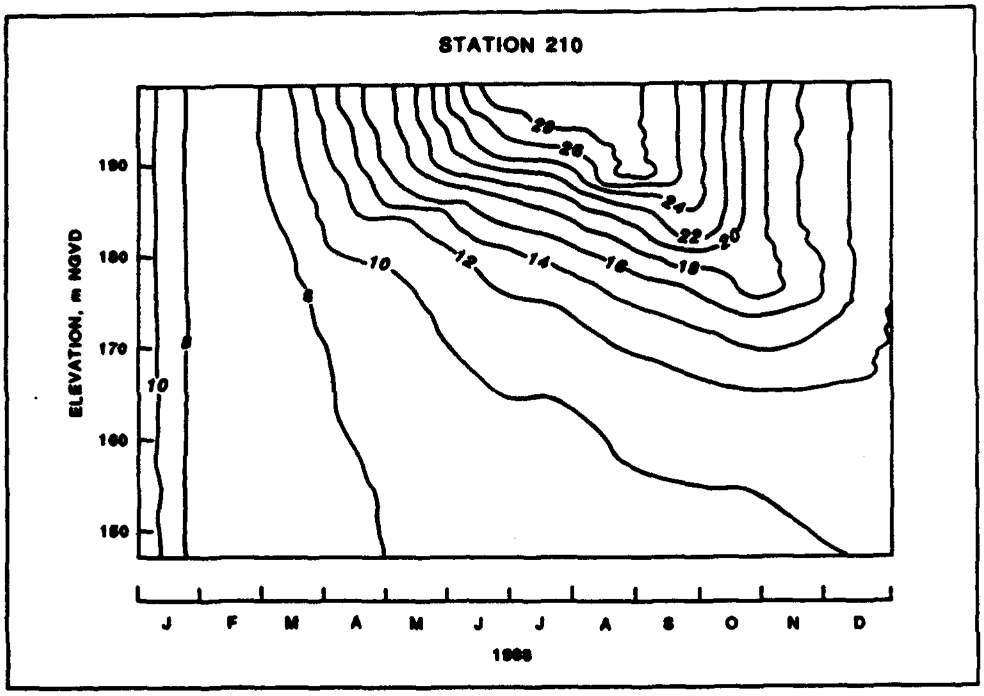

Figure 5. Temporal and vertical changes in temperature (°C) in the forebay of Hartwell Lake, sta 210

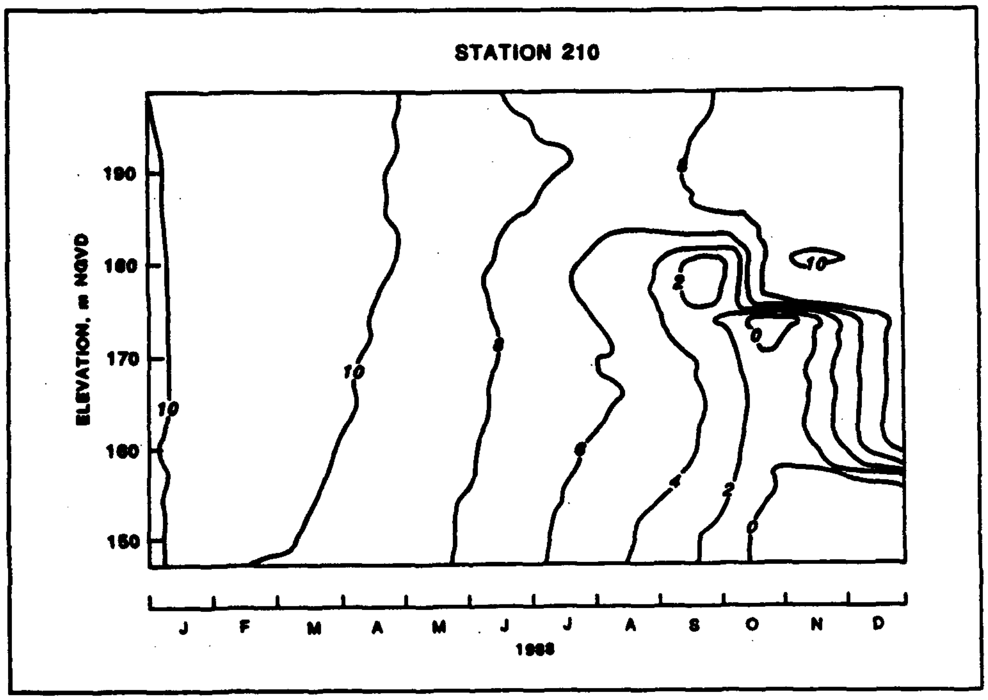

Figure 6. Temporal and vertical changes in dissolved oxygen concentrations (mg/ℓ) in the forebay of Hartwell Lake, sta 210

Seasonal measurements of chemical parameters at sta 210 indicate concentration variations were most pronounced during development of thermal gradients and anoxic conditions. Vertical gradients in manganese and iron concentrations, associated with anoxia, were the most pronounced. Increased concentrations of manganese (1.1 mg/ℓ) and iron (1.0 mg/ℓ) were observed in October in bottom waters at sta 210. However, manganese concentrations were greater than the detection limit (0.05 mg/ℓ) only during stratified conditions, at which time manganese was primarily in dissolved (i.e. reduced) forms. Iron concentrations were near 0.2 mg/ℓ for most of the year and except during anoxia, particulate (i.e. oxidized) forms comprised the majority of the total iron pool.

Seasonal trends in nitrogen concentrations were moderate during the study period, and observed concentrations were below 0.5 mg/ℓ. Minimum total nitrogen concentrations (near 0.08 mg/ℓ) were observed in January. Concentrations increased to between 0.2 and 0.4 mg/ℓ, from April to July, and decreased to 0.2 mg/ℓ in October. Dissolved nitrogen, predominantly organic forms, comprised the majority of the total nitrogen pool for most of the year. Ammonia concentrations remained near or below detection limit (0.02 mg/ℓ) except in anoxic waters where concentrations were near 0.10 mg/ℓ.

Seasonal trends in concentrations of other parameters were less apparent. Phosphorus concentrations were often below the detection limit (0.005 mg/ℓ) and observed concentrations did not exceed 0.010 mg/ℓ. Total organic carbon, concentrations of which ranged from 0.8 mg/ℓ to 1.6 mg/ℓ, was predominantly in dissolved forms. Total alkalinity values ranged from 9 to 12 mg/ℓ as $CaCO_3$ and typify low alkaline conditions of the upper Savannah basin.

Temperature and dissolved oxygen concentrations in release waters from Hartwell Dam were monitored continuously until late July, at which time the monitor became inoperable due to electrical problems. Consequently, evaluation of temperature and dissolved oxygen concentrations in the release waters is limited. Temperatures gradually increased from 8 to 10 °C (January through March) to 15 to 17 °C (observed in August and October). Dissolved oxygen concentrations remained near 11 to 8 mg/ℓ from January through June; however, a value of 3.7 mg/ℓ was observed in October, suggesting concentrations below 4 mg/ℓ occurred late in the stratified period.

Moderate seasonal trends in chemical parameter concentrations were observed in release waters and were reflective of conditions in the Hartwell Lake forebay. Maximum manganese and iron concentrations (0.27 and 0.20 mg/ℓ, respectively) were observed during periods of anoxia in the forebay. Maximum observed concentrations of total nitrogen (0.62 mg/ℓ) occurred in July. Maximum observed concentrations of total phosphorus and organic carbon (0.071 and 2.7 mg/ℓ, respectively) occurred in January and were reflective of forebay concentrations prior to the 1987 fall mixing.

Richard B. Russell Lake

Pool elevation, precipitation, mean daily inflow, and mean daily discharge for Richard B. Russell Lake during 1984-1988 are depicted in Figure 7. Although hydrologic conditions were similar to those of Hartwell Lake, pool elevations in Richard B. Russell Lake were maintained at near-normal levels during 1988, due to design drawdown of the conservation pool.

Spatial patterns in thermal structure were observed along the main stem of Richard B. Russell Lake (Figure 8). Thermal stratification was present from the dam to the headwater region (sta 060B to 190) from late March through September. Changes in monthly temperature at sta 120 exemplify the seasonal pattern of thermal development in the lake (Figure 9). Stratification began in late March and a well-established thermocline was present near a depth of 6 m by mid-May. The thermocline remained near a depth of 6 to 8 m throughout the season. During most of thermal stratification, epilimnetic temperatures were between 20 and 28 °C and hypolimnetic temperatures were between 12 and 16 °C. Seasonal cooling in late September and early October decreased surface water temperatures and gradually weakened the thermocline. Continued cooling and wind-induced mixing resulted in complete mixing, and nearly isothermal conditions were observed in late October.

Temporal and spatial gradients in dissolved oxygen were apparent along the main stem of the lake during the period of stratification (Figure 10). Dissolved oxygen concentrations remained near 8 to 10 mg/ℓ in the epilimnion of the lake, while gradually declining in the hypolimnion. Dissolved oxygen concentrations greater than 4 mg/ℓ were present throughout surface waters in the main stem and throughout the water column in the mid- and upstream region of the main stem. Anoxic conditions in the main stem of the lake were confined to bottom waters between sta 100 and sta 120. Data from sta 120 indicate that anoxic conditions in the hypolimnion were established by mid-June and continued until late October (Figure 11). In general, anoxic conditions were confined to the bottom 6 to 10 m in the downstream region of the lake.

Gradients in conductivity values observed in the main stem of the lake coincided with anoxic conditions (Figure 12). Conductivity levels were near 40 µS for most of the lake; however, increased levels (40 to 70 µS) were observed in anoxic waters of the downstream region. Increased conductivity may be attributable to increased concentrations of dissolved constituents released from the sediments in conjunction with anaerobic processes.

Spatial trends in the concentrations of chemical parameters were most pronounced during stratification. Maximum observed main stem concentrations occurred in the bottom waters of the downstream region of the lake coincident with anoxic conditions. As a result, longitudinal and vertical concentration gradients were apparent along the main stem of the lake (Figure 13).

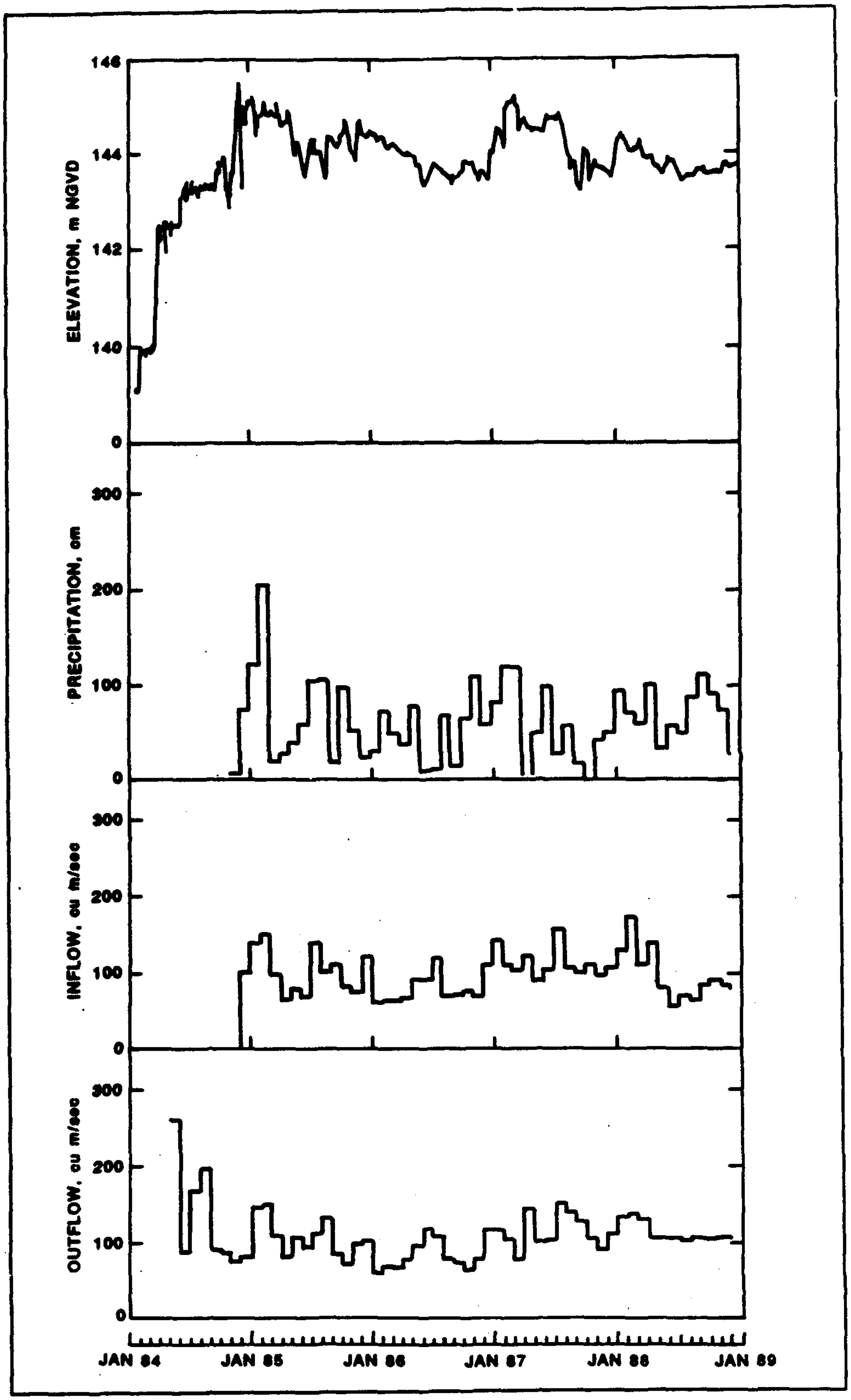

Figure 7. Pool elevation, precipitation, inflows, and outflows for Richard B. Russell Lake, 1984-1988

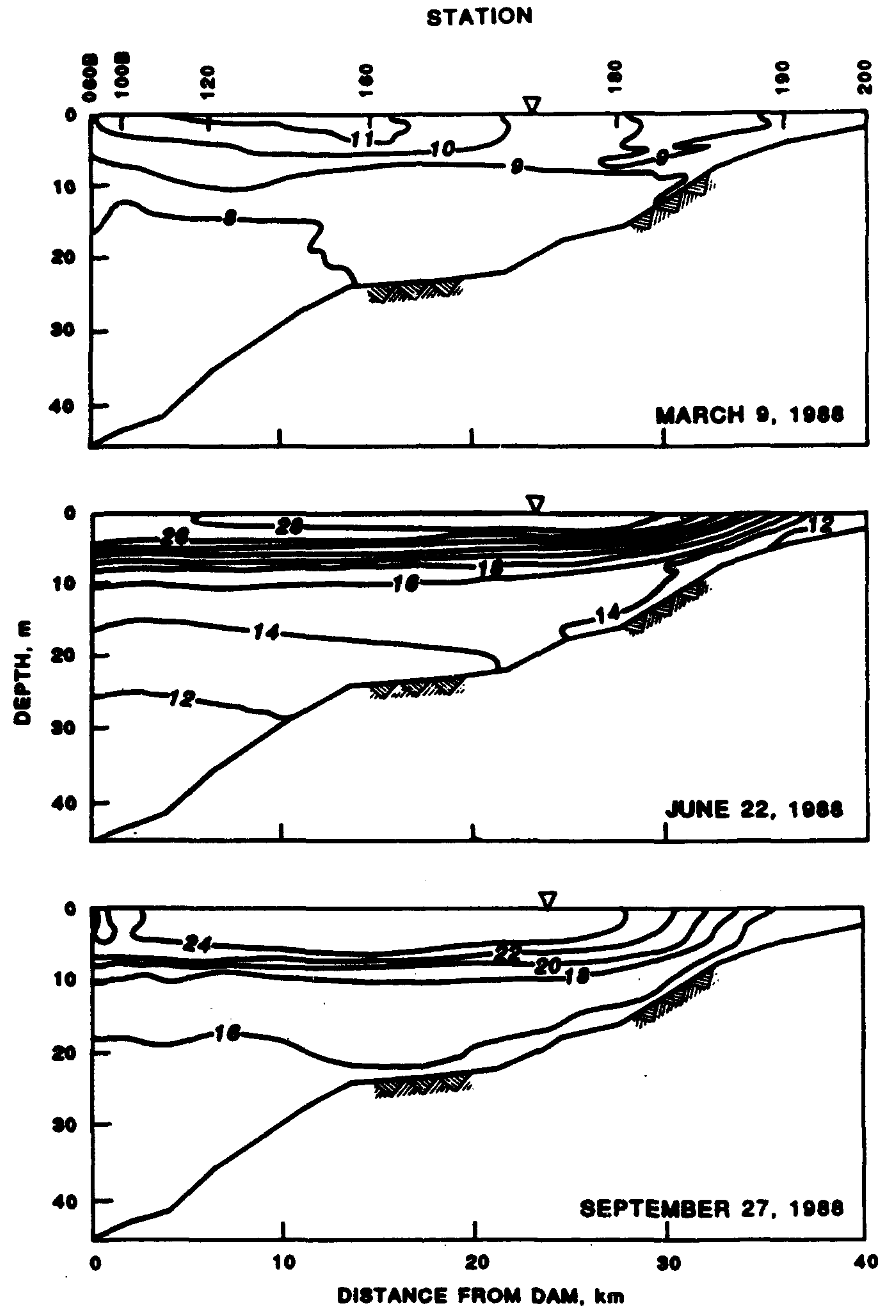

Figure 8. Patterns of spatial distribution of temperatures (°C) in the main stem of Richard B. Russell Lake, March, June, and September 1988

Total iron concentrations ranged from detection limit (0.05 mg/ℓ) to greater than 6.0 mg/ℓ; however, concentrations greater than 0.5 mg/ℓ were confined to depths greater than 25 m. Temporal and vertical development of

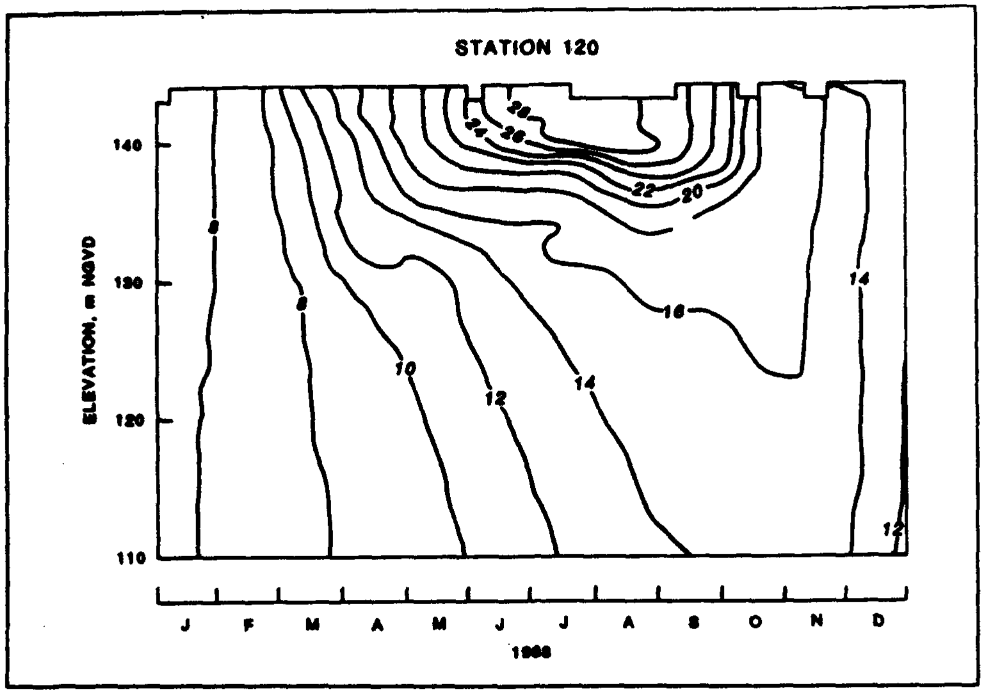

Figure 9. Temporal and vertical changes in temperature (°C) in the lower region of Richard B. Russell Lake, sta 120

concentration gradients depict spatial variability in the downstream region of the lake (Figure 14). In general, concentrations increased in a downstream direction to sta 100B; however, concentrations downstream of sta 100B were lower during the study period. Iron was primarily in dissolved forms in anoxic waters (i.e. sta 120 hypolimnion and immediate bottom waters of sta 100B). Increased particulate iron concentrations in areas near and downstream of the oxygenation system suggest oxidation and formation of particulate iron in the region of the lake influenced by hypolimnetic oxygenation (Figure 15). Resultant precipitation of particulate iron downstream of the oxygenation system may account for decreased total iron concentrations observed in the forebay of the lake.

Total manganese concentrations ranged from detection limit (0.05 mg/ℓ) to 1.56 mg/ℓ (Figure 13) and dissolved manganese comprised the majority of the total manganese pool. Total manganese concentrations began to increase in bottom waters in late April, and peak concentrations were observed from July through October (Figure 16). Concentrations returned to near detection limits following the fall mixing. Concentrations greater than 0.50 mg/ℓ were confined to depths greater than 25 m. Particulate manganese was not observed in the lake at concentrations greater than 0.1 mg/ℓ during the study period.

Total phosphorus and nitrogen concentrations displayed temporal and spatial patterns similar to iron and manganese (see Figure 13), but concentration gradients were less pronounced. Generally, total phosphorus concentrations ranged from 0.01 mg/ℓ to 0.04 mg/ℓ and concentrations greater than

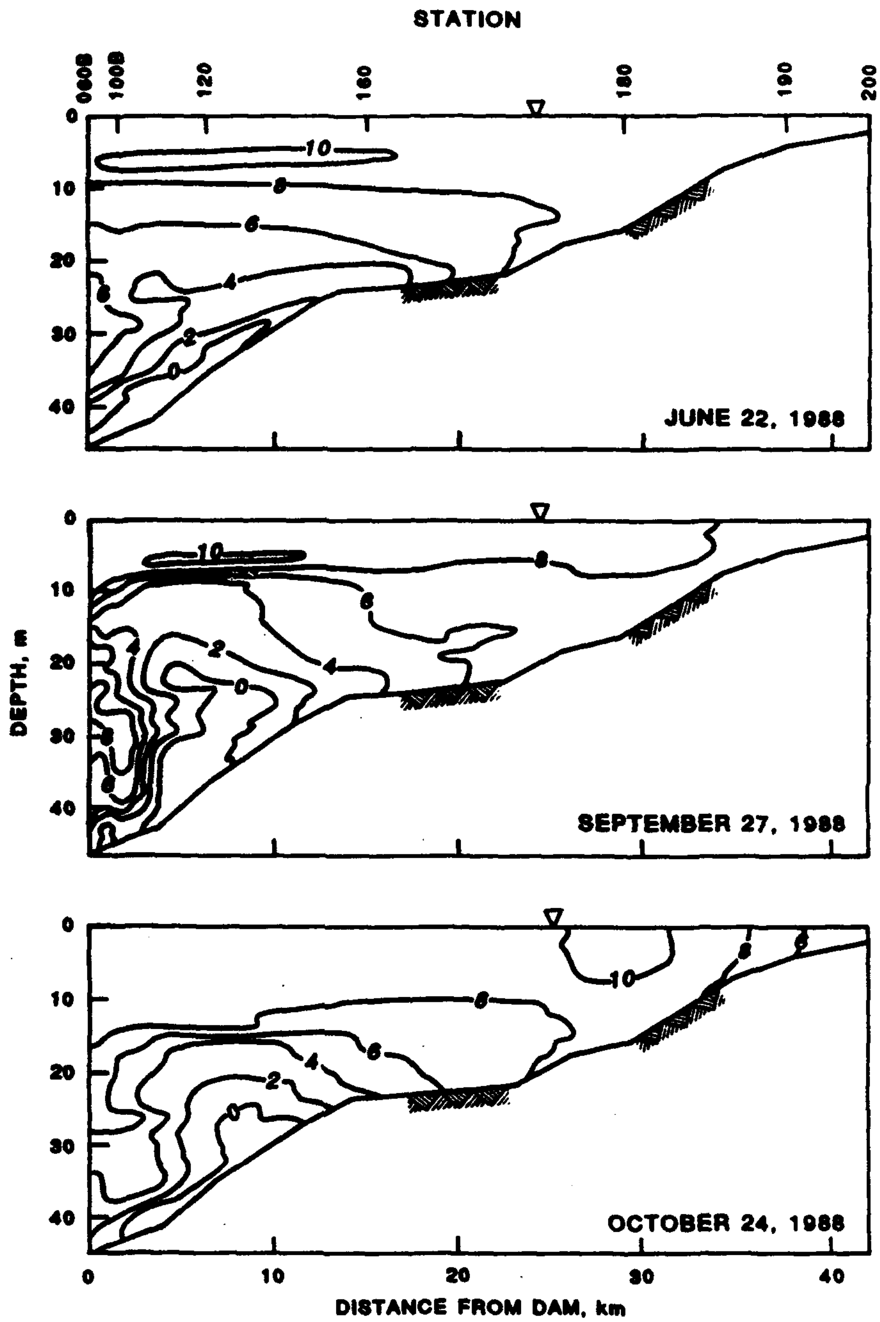

Figure 10. Patterns of spatial distribution of dissolved oxygen concentrations (mg/ℓ) in the main stem of Richard B. Russell Lake, June, September, and October 1988

0.02 mg/ℓ were confined to the bottom 10 m of the downstream region of the lake. Similarly, total nitrogen ranged from 0.40 mg/ℓ to 0.80 mg/ℓ and concentrations greater than 0.6 mg/ℓ were confined to bottom depths in the

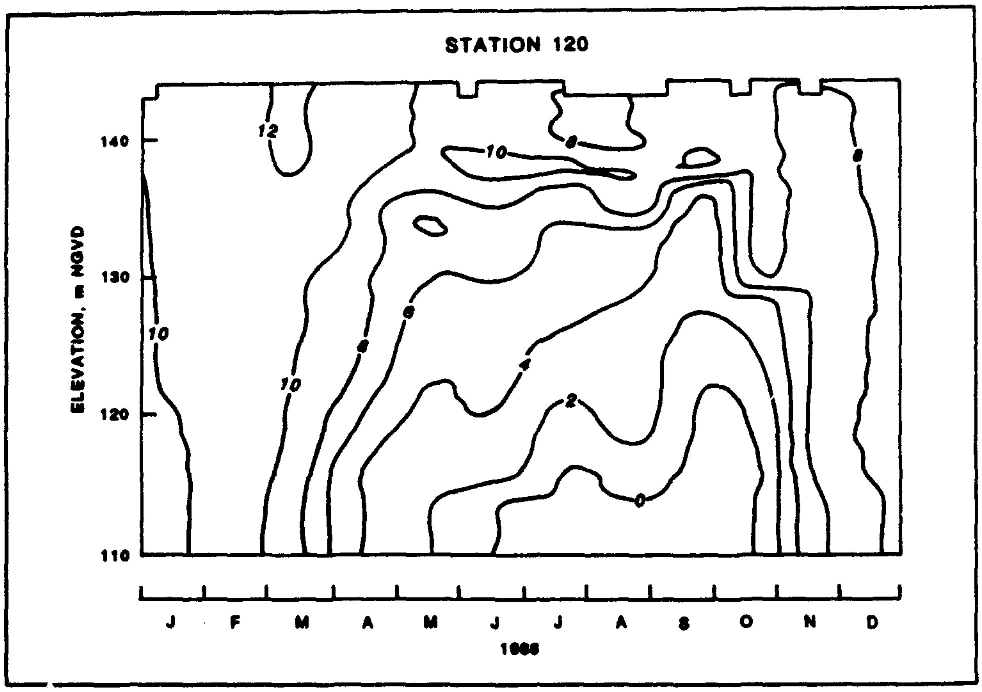

Figure 11. Temporal and vertical changes in dissolved oxygen concentrations (mg/ℓ) in the lower region of Richard B. Russell Lake, sta 120

downstream region of the lake. Peak concentrations of total phosphorus (0.118 mg/ℓ) and total nitrogen (1.13 mg/ℓ) were observed in bottom waters at sta 100B during anoxic conditions in August and September.

While seasonal development of hypolimnetic anoxia was similar to that of the main stem of the lake, the magnitude of anoxic conditions in the tributary embayments was greater (i.e. sta 130, Figure 17). Additionally, maximum observed concentrations of manganese, iron, carbon, nitrogen, and phosphorus occurred at embayment stations (sta 130 and 140; Table 3).

Temperature and dissolved oxygen concentrations in release waters from Richard B. Russell Dam (i.e. sta 050, Figure 18) displayed seasonal trends reflective of changing conditions in the Richard B. Russell Lake forebay. Temperatures gradually increased from approximately 8 to 10 °C during February through March to 15 to 17 °C during September through November. Dissolved oxygen concentrations, near 8 to 11 mg/ℓ from January through April, were maintained near 6 mg/ℓ from May through mid-November, due to operation of the oxygenation system. Coincident with fall mixing in Richard B. Russell Lake, dissolved oxygen concentrations gradually returned to near 8 to 12 mg/ℓ during November and December.

Moderate seasonal trends in chemical parameter concentrations were observed in release waters and were reflective of stratified conditions in mid-depth waters of the Richard B. Russell Lake forebay region. Manganese and iron concentrations ranged from 0.2 to 0.3 and from 0.2 to 0.4 mg/ℓ,

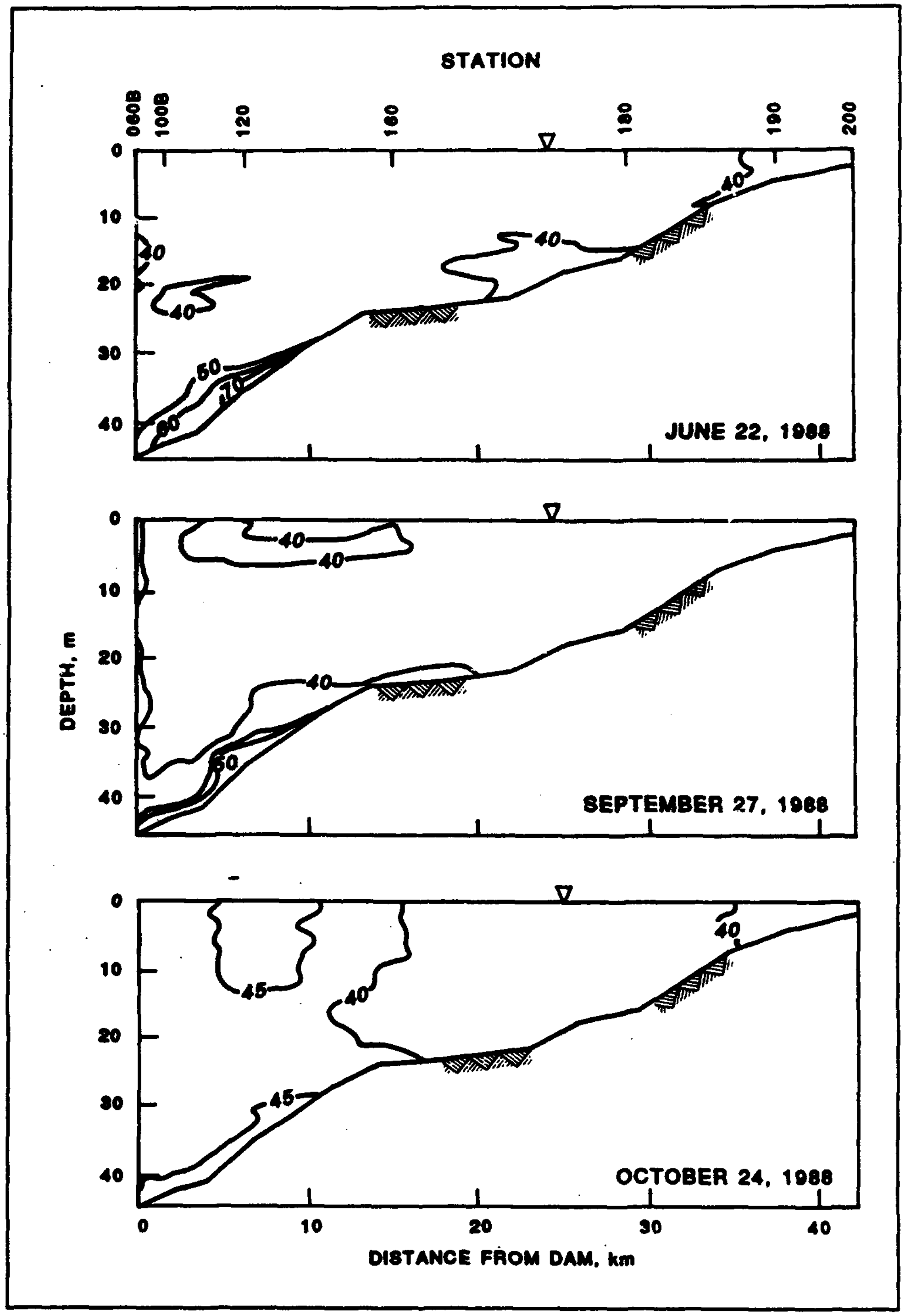

Figure 12. Patterns of spatial distribution of conductivity (µS) in the main stem of Richard B. Russell Lake, June, September, and October 1988

respectively, from June through November. In general, dissolved manganese and particulate iron comprised the majority of the total manganese and iron pools, respectively, in the release waters.

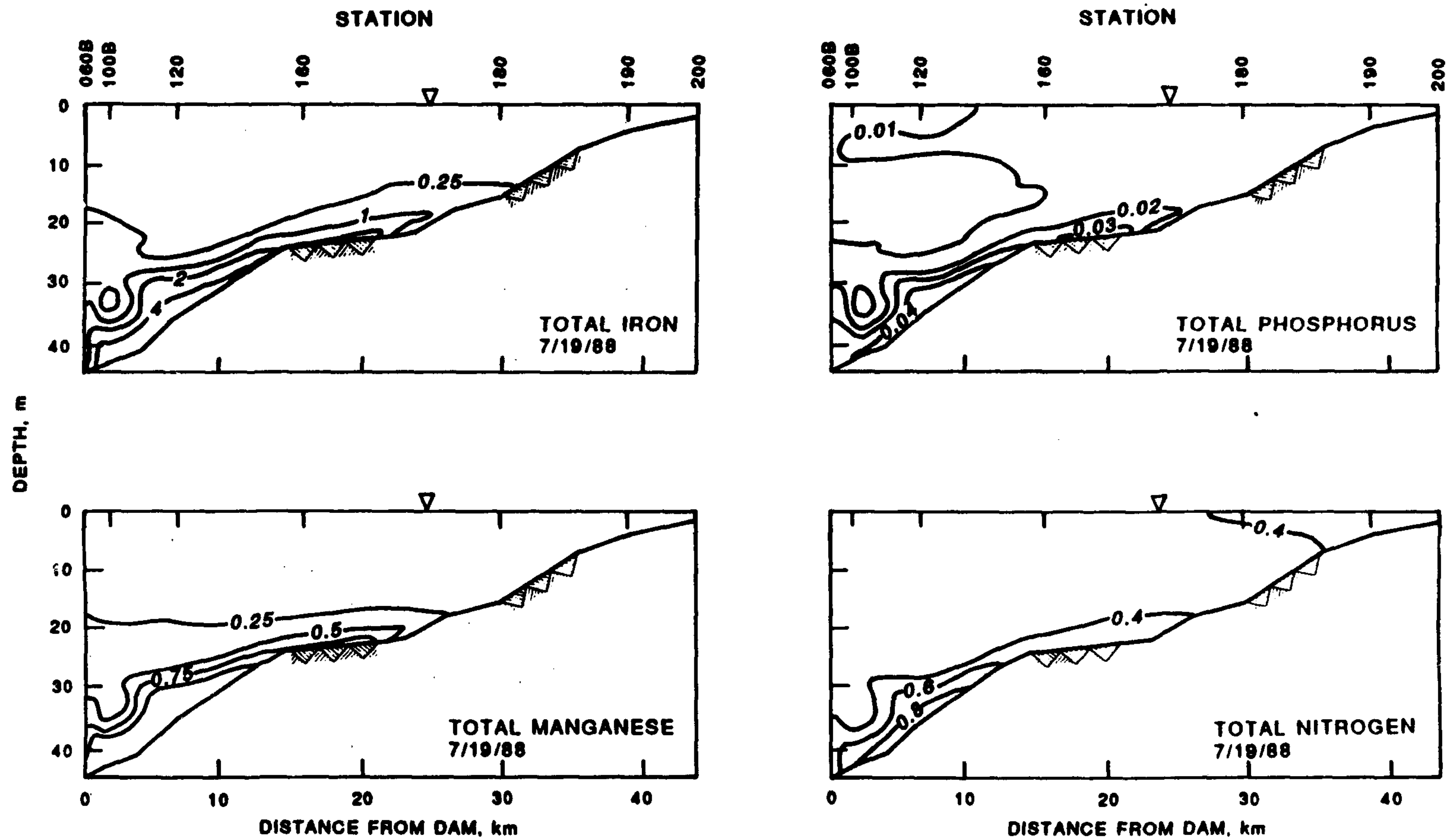

Figure 13. Spatial distribution of total iron, manganese, phosphorus, and nitrogen concentrations (mg/ℓ) in the main stem of Richard B. Russell Lake, July 1988

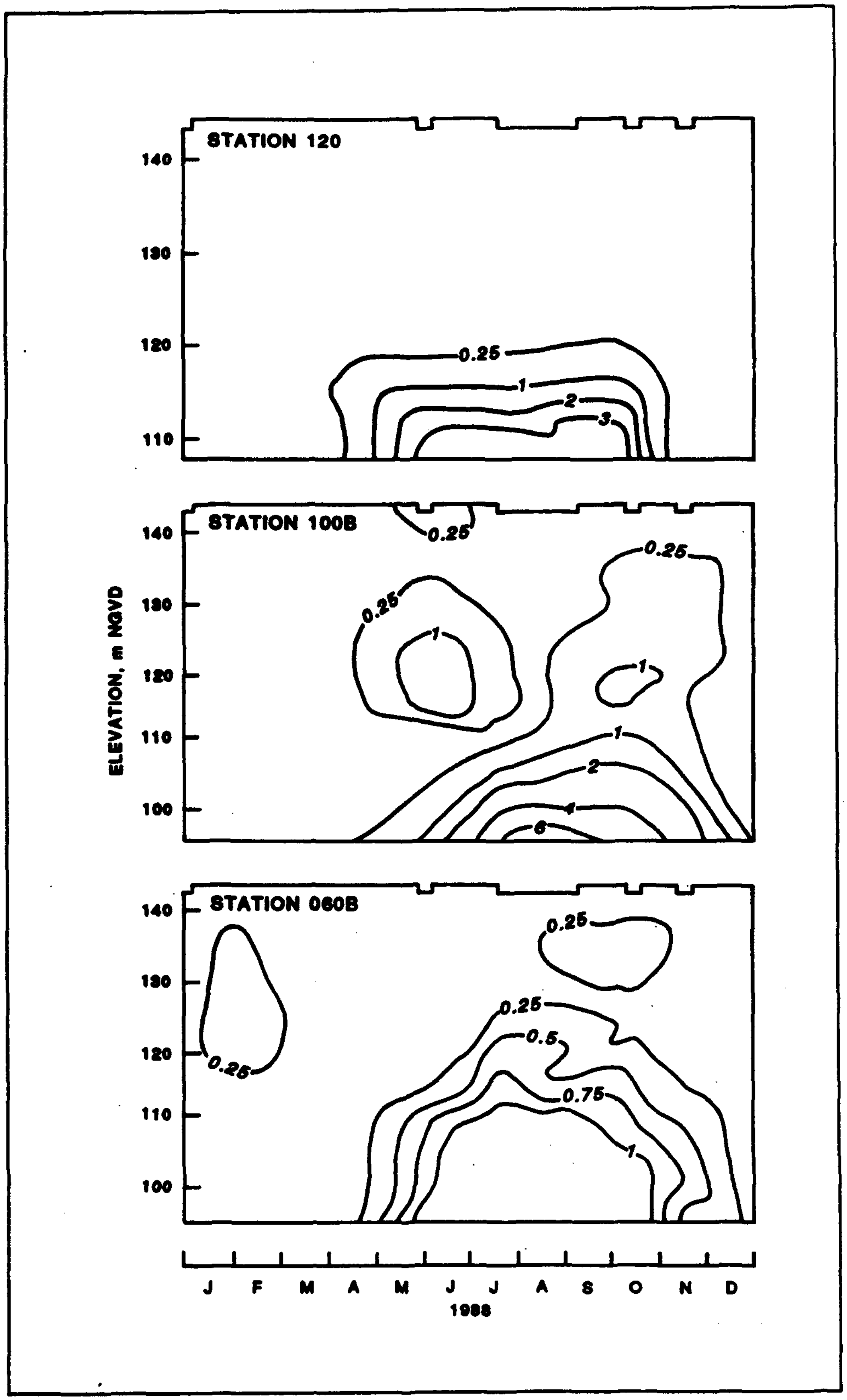

Figure 14. Temporal and vertical changes in total iron concentrations (mg/ℓ) in the forebay and lower region of Richard B. Russell Lake

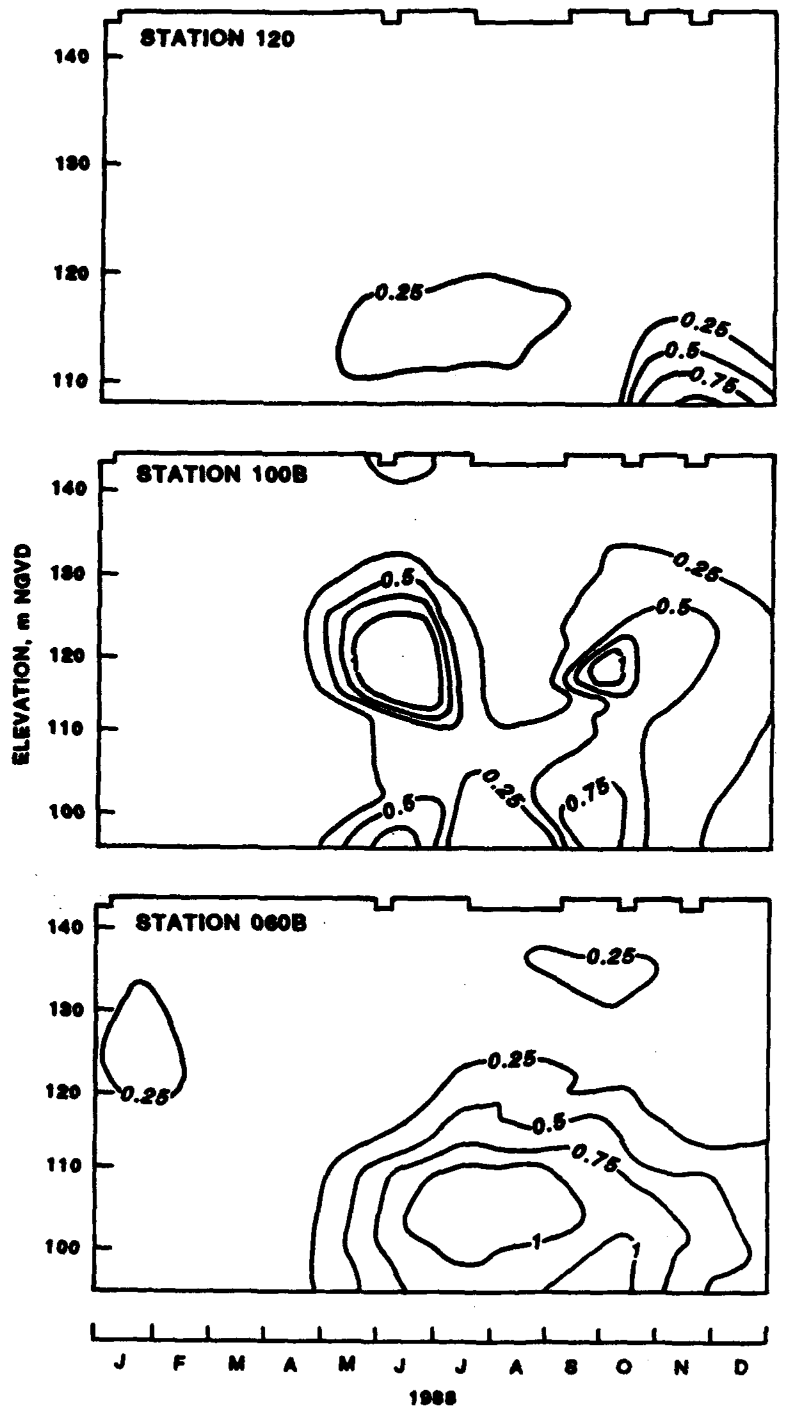

Figure 15. Temporal and vertical changes in particulate iron concentrations (mg/ℓ) in the forebay and lower region of Richard B. Russell Lake

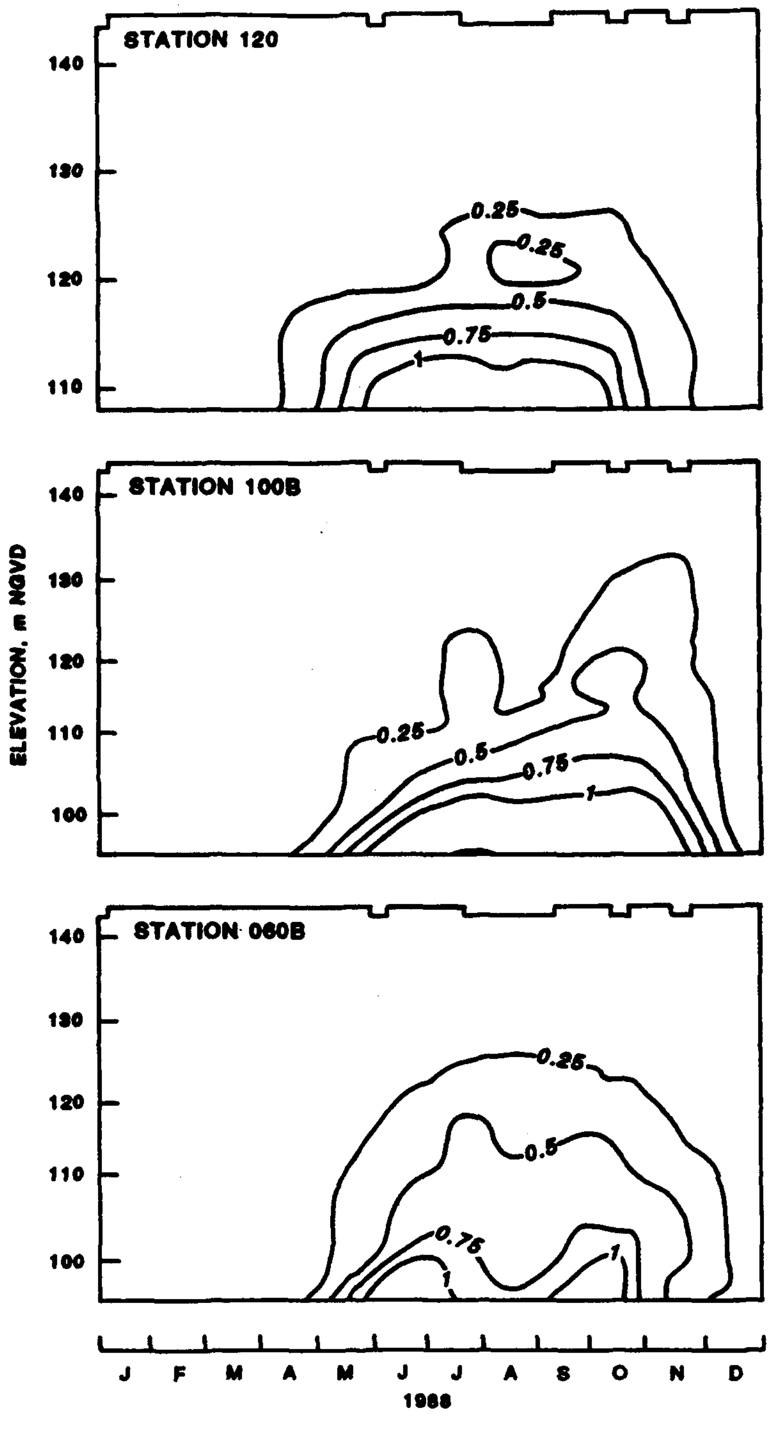

Figure 16. Temporal and vertical changes in total manganese concentrations (mg/l) in the forebay and lower region of Richard B. Russell Lake

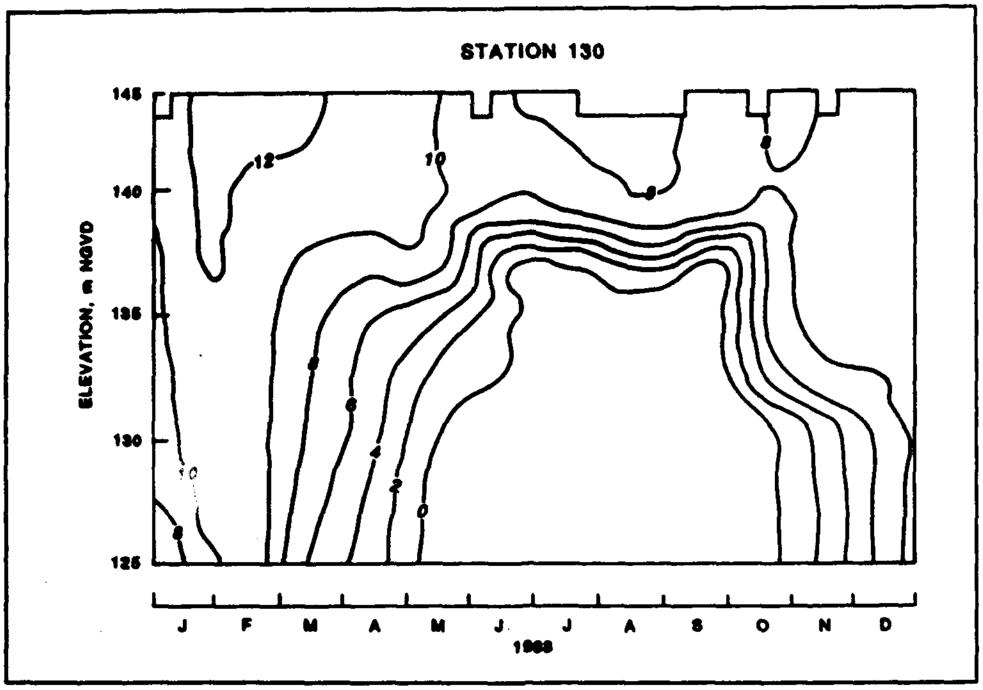

Figure 17. Temporal and vertical changes in dissolved oxygen concentrations (mg/ℓ) in the Beaverdam Creek embayment, sta 130

Table 3
Maximum Observed Concentrations (mg l^{-1}) of Chemical Parameters in Richard B. Russell Lake, 1988

Station	Total Manganese	Total Iron	Total Organic Carbon	Total Nitrogen	Total Phosphorus
Tributaries					
130	2.24*	12.75*	4.4	2.13	0.103
140	2.00	6.71	7.3*	2.91*	0.206*
150	0.89	1.95	4.2	0.96	0.049
Main Stem					
060B	1.54	2.70	2.3	0.72	0.029
100B	2.06	7.16	5.5	1.18	0.118
120	1.56	4.54	2.3	0.97	0.050
160	0.31	0.54	2.0	0.46	0.022
180	0.17	0.56	1.8	0.42	0.024

* Denotes maximum values observed in 1988.

Seasonal trends for other parameters in release waters were less apparent. Maximum nitrogen concentrations were observed in the spring and summer and ranged from 0.86 to 0.76 mg/ℓ (April and July, respectively). Minimum nitrogen concentrations occurred following the fall mixing and during winter

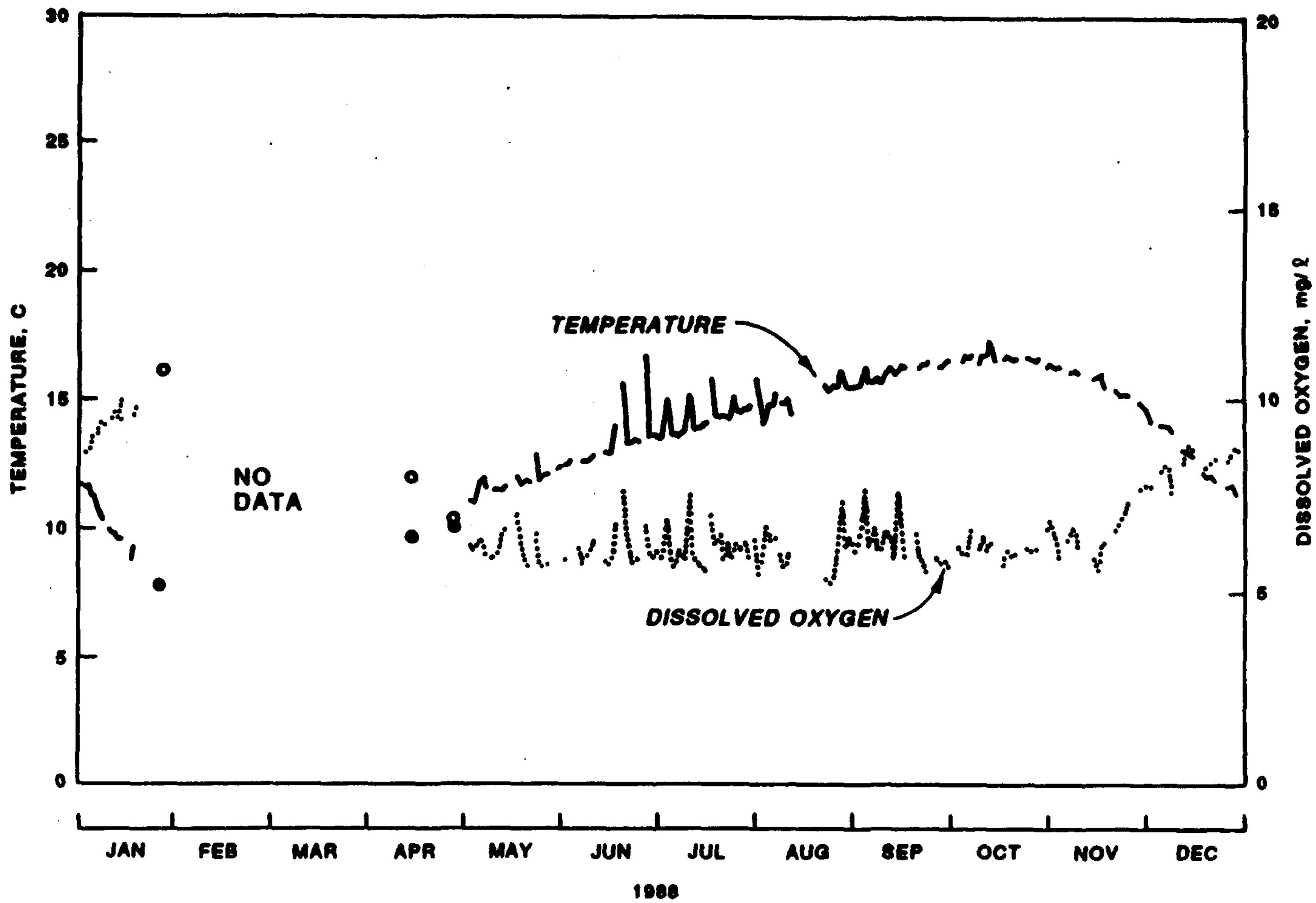

Figure 18. Temperatures and dissolved oxygen concentrations in the Richard B. Russell Dam releases, sta 050, 1988

and ranged from 0.46 to 0.29 mg/ℓ (October and February, respectively). Dissolved forms of nitrogen comprised the majority of the total nitrogen pool in release waters at all times. Observed values of conductivity, pH, alkalinity, carbon, and phosphorus were similar to values in the forebay and seasonal trends were less pronounced.

J. Strom Thurmond Lake

Pool elevation, monthly precipitation, mean daily inflow, and mean daily discharge for J. Strom Thurmond Lake during 1984-1988 are depicted in Figure 19. While pool elevations returned to near-normal levels early in 1988, decreased precipitation and inflow levels and operational requirements resulted in low pool elevations from June through December.

Spatial patterns in thermal structure were observed temporally along the main stem of J. Strom Thurmond Lake (Figure 20). Thermal stratification was present from J. Strom Thurmond Dam to the headwater region (sta 020 to 050) from April through September. Monthly temperature data from sta 020, located in the downstream region of J. Strom Thurmond Lake, depict the temporal pattern of thermal development in the lake (Figure 21). Thermal stratification began in late April with a well established thermocline near 6 to 8 m observed by mid-May. The thermocline remained near a depth of 8 to 10 m throughout the season. Temperatures in the epilimnion were between 24 and 30 °C and hypolimnetic temperatures were between 14 and 18 °C, during most of the period of stratification. Seasonal cooling in late September and early October reduced surface water temperatures and gradually weakened the thermocline. Continued cooling of surface waters resulted in near isothermal conditions by mid-October.

Temporal and spatial gradients in dissolved oxygen were apparent along the main stem of the lake during the period of stratification (Figure 22). Dissolved oxygen concentrations, while remaining near 8 to 10 mg/ℓ in the epilimnion of the lake, gradually declined in the hypolimnion in the downstream region of the lake. Anoxic conditions in the hypolimnion of the downstream region of the lake (i.e. sta 020) were established by mid- to late August and continued until late October and were present to within 10 m from the surface (Figure 23 top). Conversely, dissolved oxygen concentrations in the midstream region of the lake did not decline below 4.0 mg/ℓ (Figure 23 bottom).

Increases in conductivity were observed along the main stem of the lake coincident with the development of anoxic conditions (Figure 24). While conductivity levels were between 40 and 45 µS for most of the year, levels in the anoxic waters ranged from 50 to 60 µS, due to increased concentrations of dissolved constituents.

Spatial trends in the concentrations of chemical parameters were most pronounced during stratification. As a result, longitudinal and vertical concentration gradients were apparent along the main stem of the lake (Figure 25).

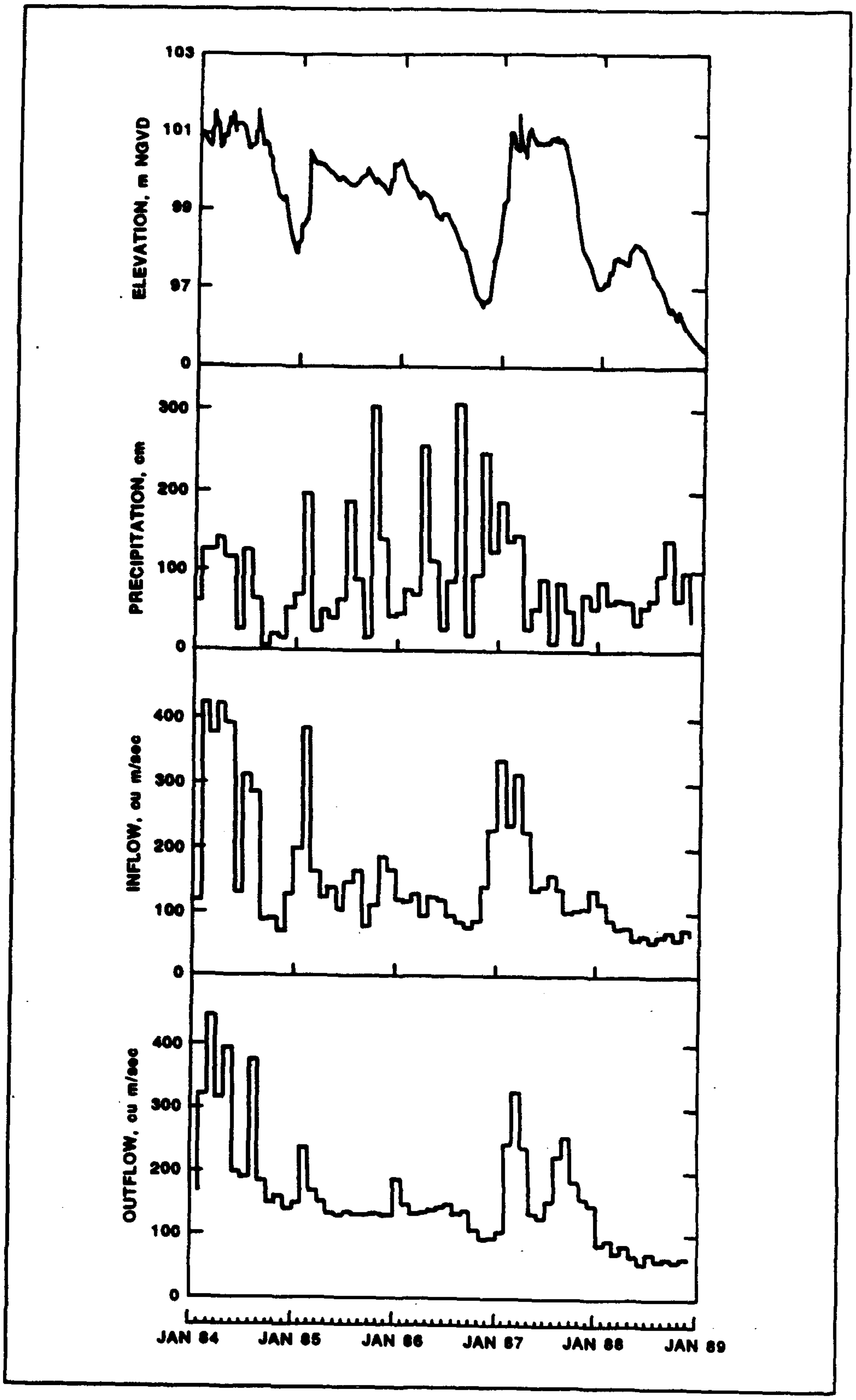

Figure 19. Pool elevation, precipitation, inflows, and outflows for J. Strom Thurmond Lake, 1984-1988

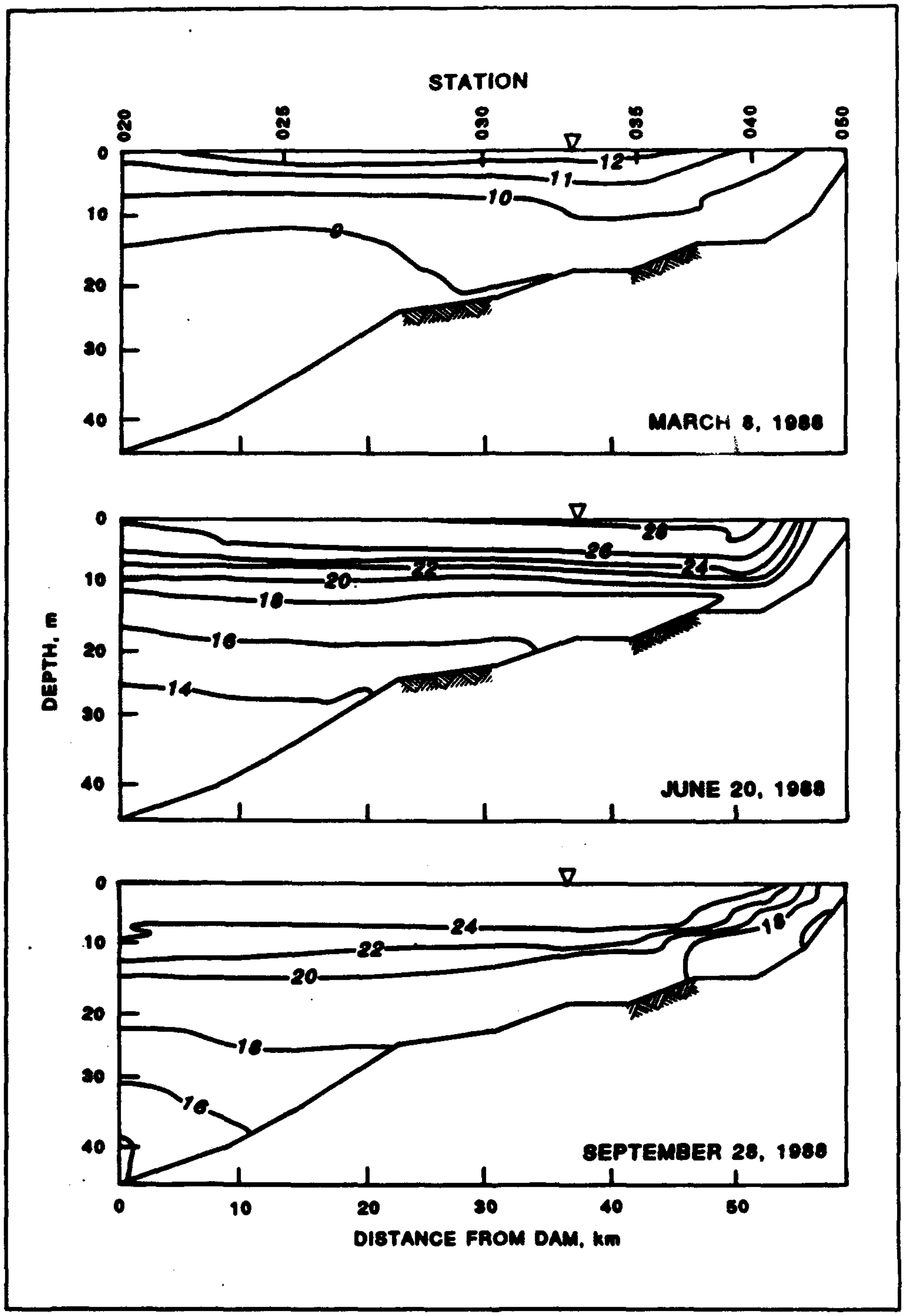

Figure 20. Patterns of spatial distribution of temperatures (°C) in the main stem of J. Strom Thurmond Lake, March, June, and September 1988

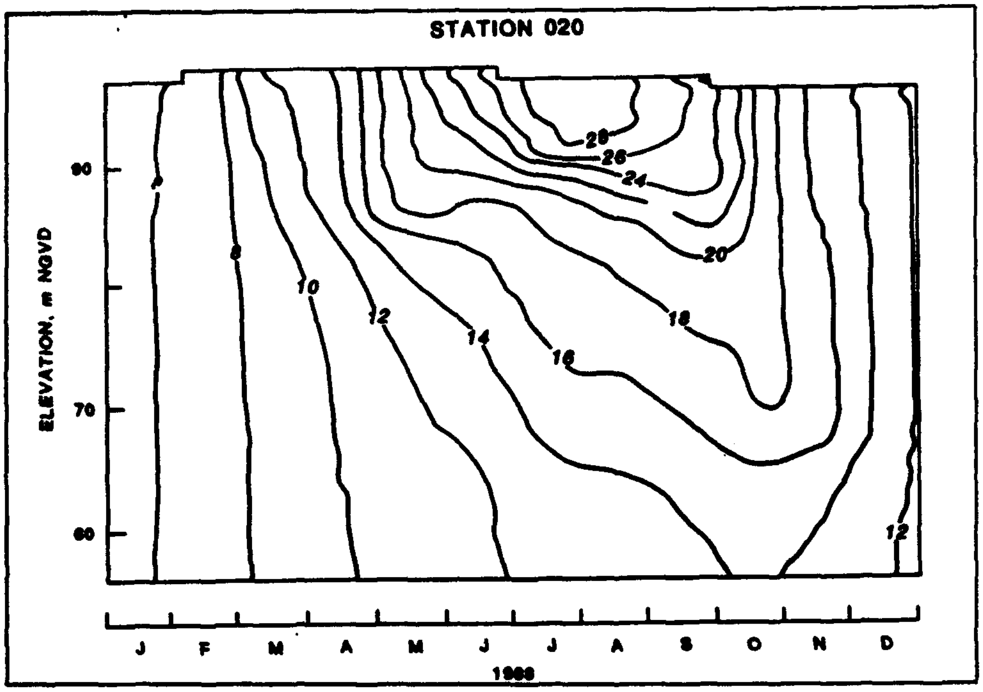

Figure 21. Temporal and vertical changes in temperature (°C) in the forebay of J. Strom Thurmond Lake, sta 020

Varied spatial distributions of iron and manganese concentrations were observed, however, during anoxia in the downstream region of the lake (Figure 26).

Total iron concentrations ranged from the detection limit (0.05 mg/ℓ) to 1.59 mg/ℓ, and maximum iron concentrations observed in the main stem occurred at sta 030. Concentrations in the downstream region of the lake were below 0.5 mg/ℓ, even during anoxia. Particulate iron comprised the majority of the total iron pool at all times.

Conversely, total manganese concentrations in the main stem ranged from detection limit (0.05 mg/ℓ) to 5.10 mg/ℓ and maximum observed concentrations occurred in the downstream region of the lake, i.e. sta 020 (Figure 26). However, pronounced concentration gradients were confined to depths greater than 30 m. Additionally, dissolved manganese comprised the majority of the total manganese pool at all times and particulate manganese concentrations greater than 0.1 mg/ℓ were not observed during the study period.

Seasonal trends in the distribution of manganese were most apparent during stratification, coincident with anoxia (Figure 27). Concentrations of dissolved manganese began to increase in bottom waters in July, were at peak concentrations from July through early October, and returned to levels near the detection limit (0.05 mg/ℓ) following fall mixing.

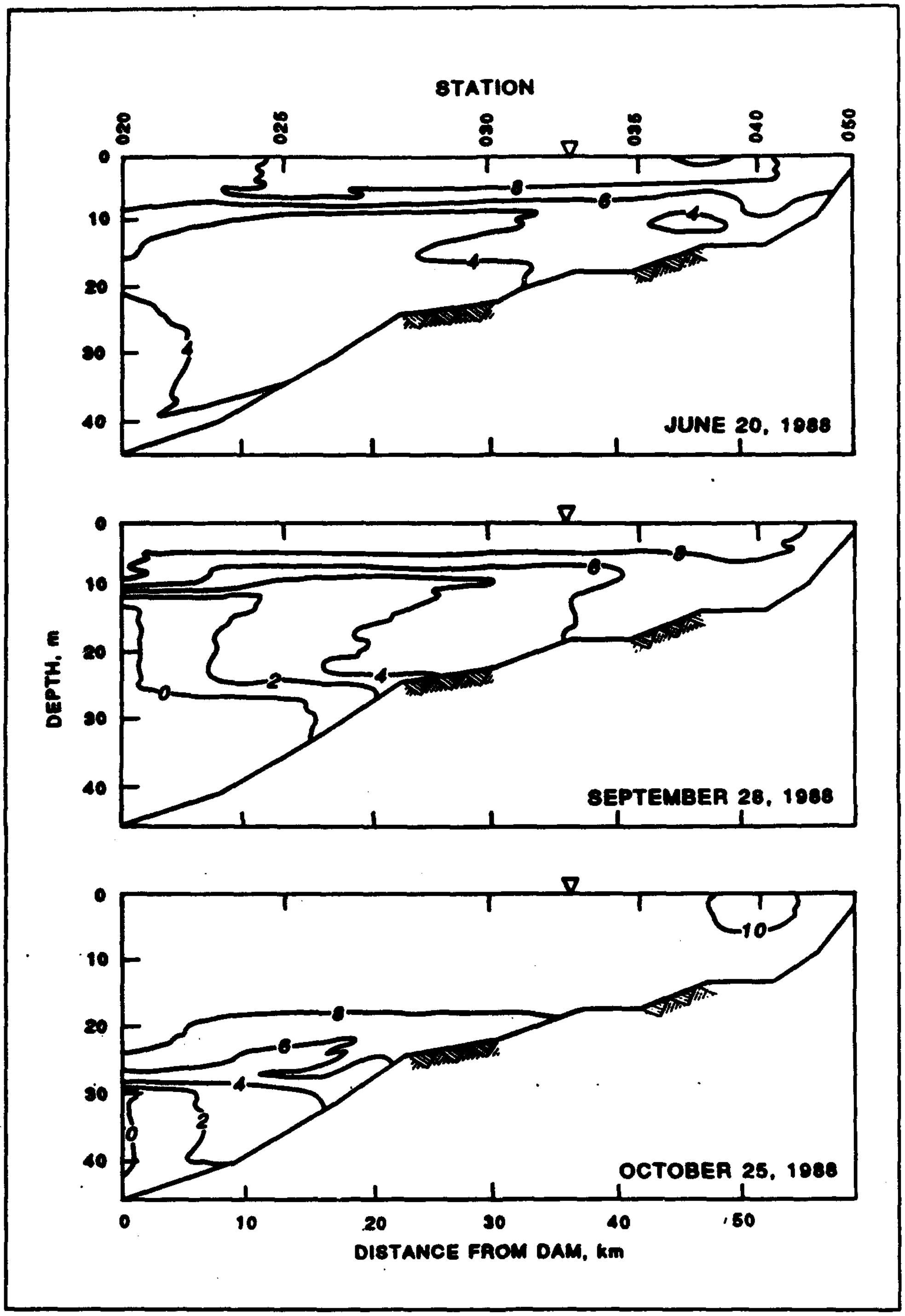

Figure 22. Patterns of spatial distribution of dissolved oxygen concentrations (mg/ℓ) in the main stem of J. Strom Thurmond Lake, June, September, and October 1988

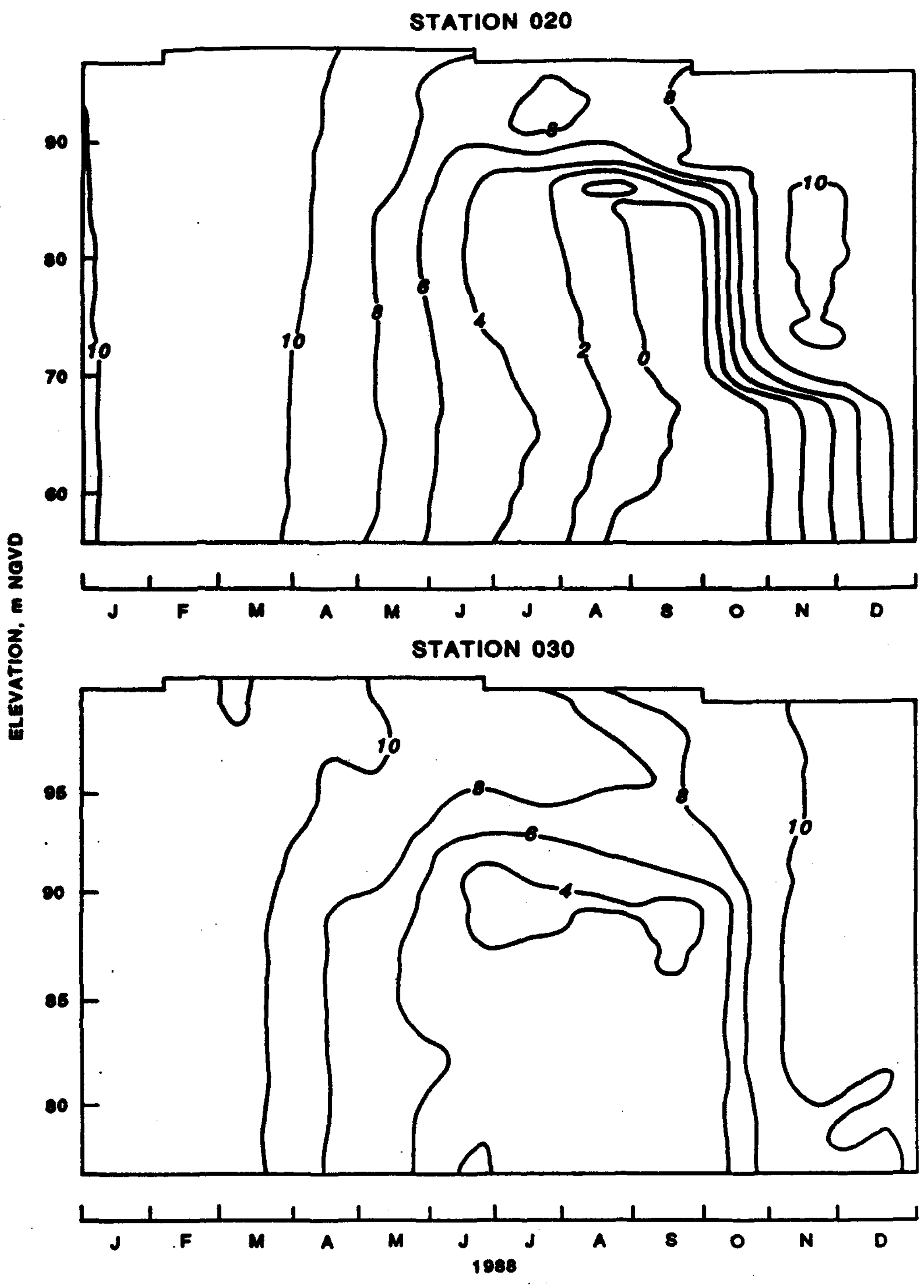

Figure 23. Temporal and vertical changes in dissolved oxygen concentrations (mg/ℓ) in the forebay, sta 020, and midlake region, sta 030, of J. Strom Thurmond Lake

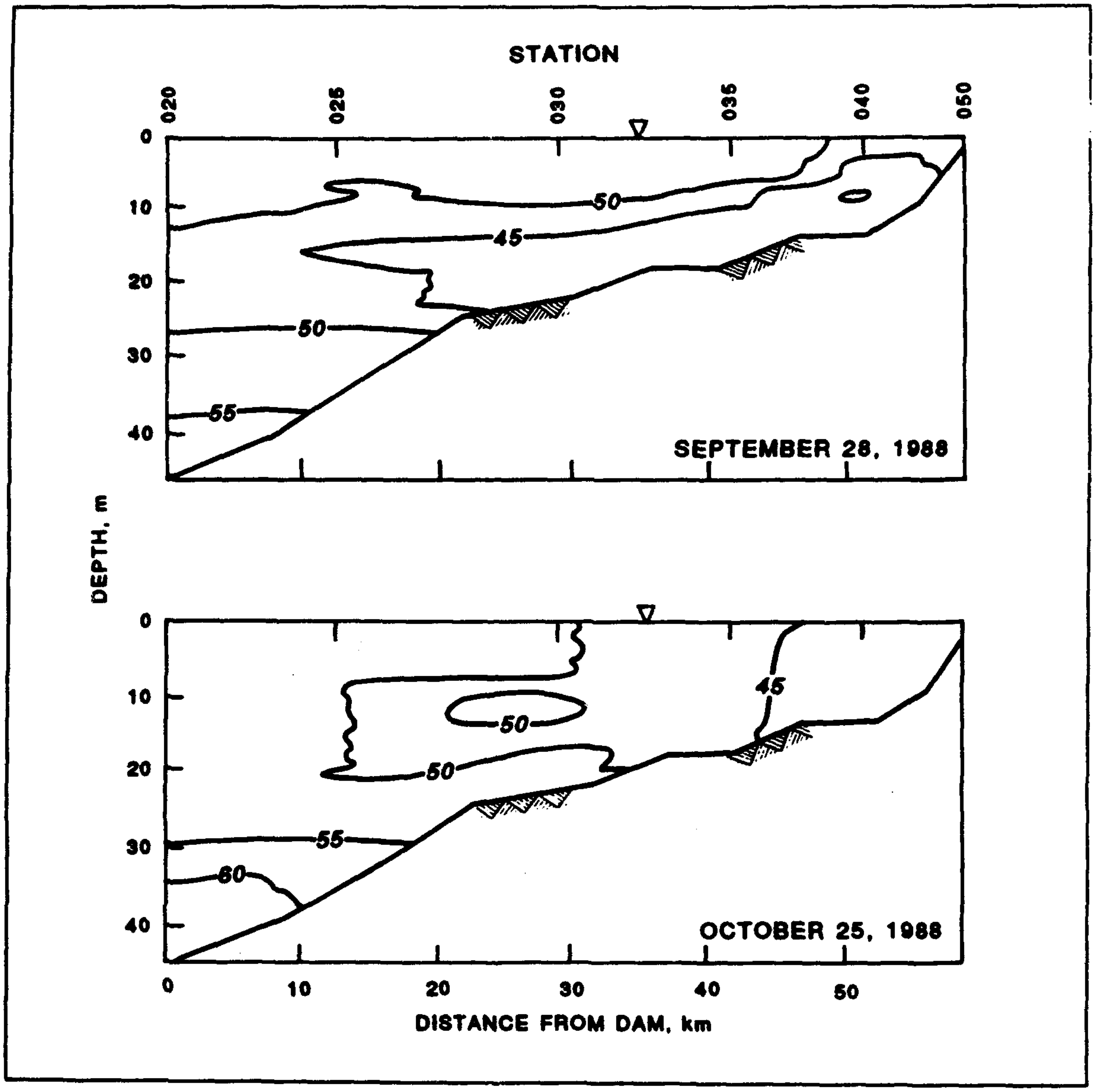

Figure 24. Patterns of spatial distribution of specific conductance (μS) in the main stem of J. Strom Thurmond Lake, September and October, 1988

Concentrations of total phosphorus and nitrogen ranged from 0.005 mg/ℓ to 0.034 mg/ℓ and 0.1 to 0.7 mg/ℓ, respectively, and pronounced gradients were not observed in the main stem of the lake (see Figure 25). Seasonal variation was most pronounced for nitrogen concentrations with maximum values observed in April and July, while minimum values occurred in February and October. Total organic carbon ranged from 1 to 3 mg/ℓ and temporal and spatial trends were not apparent.

Maximum observed concentrations of total iron (3.1 mg/ℓ), organic carbon (4.1 mg/ℓ), nitrogen (1.2 mg/ℓ), and phosphorus (0.078 mg/ℓ) occurred in embayments (i.e. sta 029 or 015). Additionally, increased concentrations of

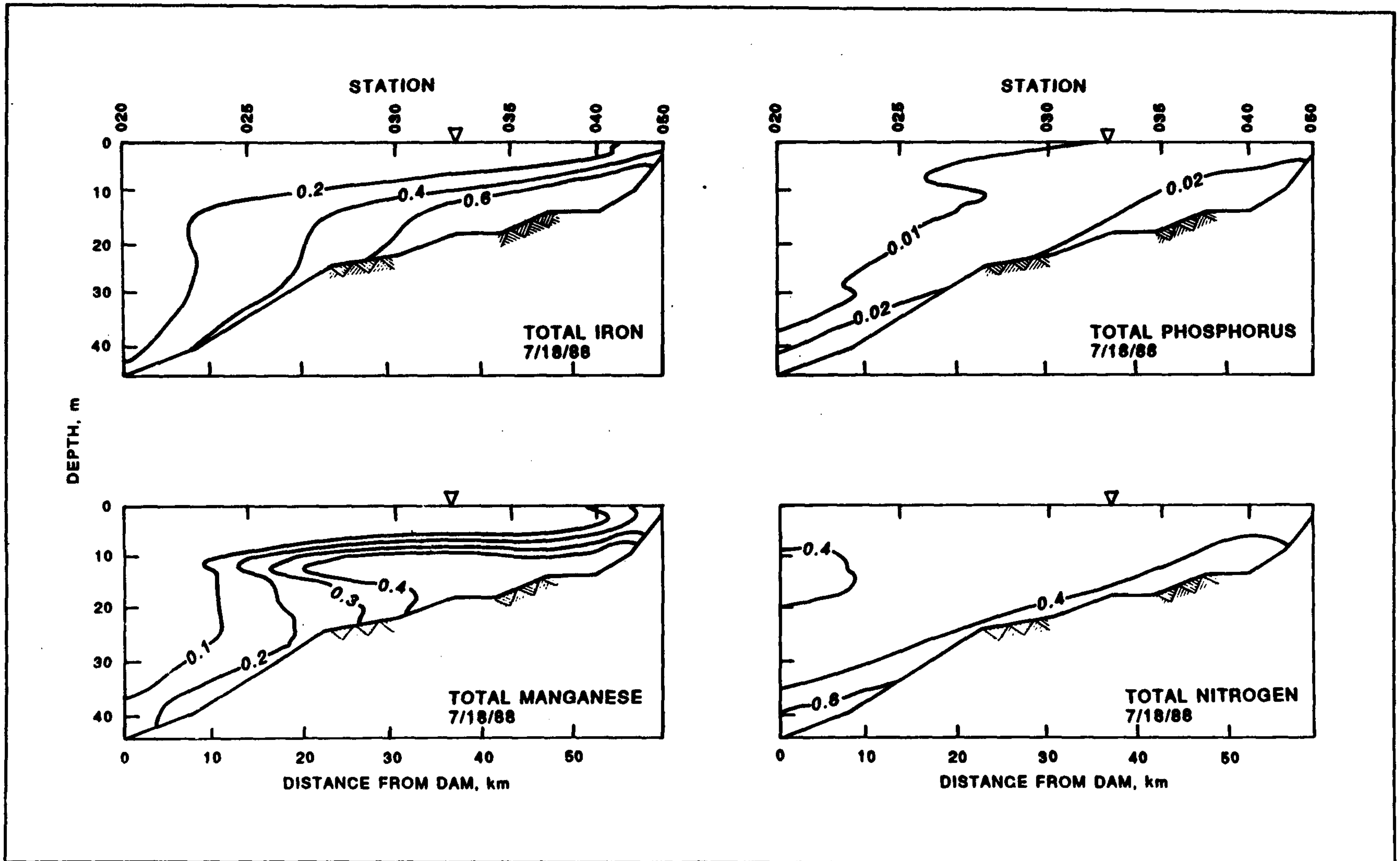

Figure 25. Spatial distribution of total iron, manganese, phosphorus, and nitrogen concentrations (mg/ℓ) in the main stem of J. Strom Thurmond Lake, July 1988

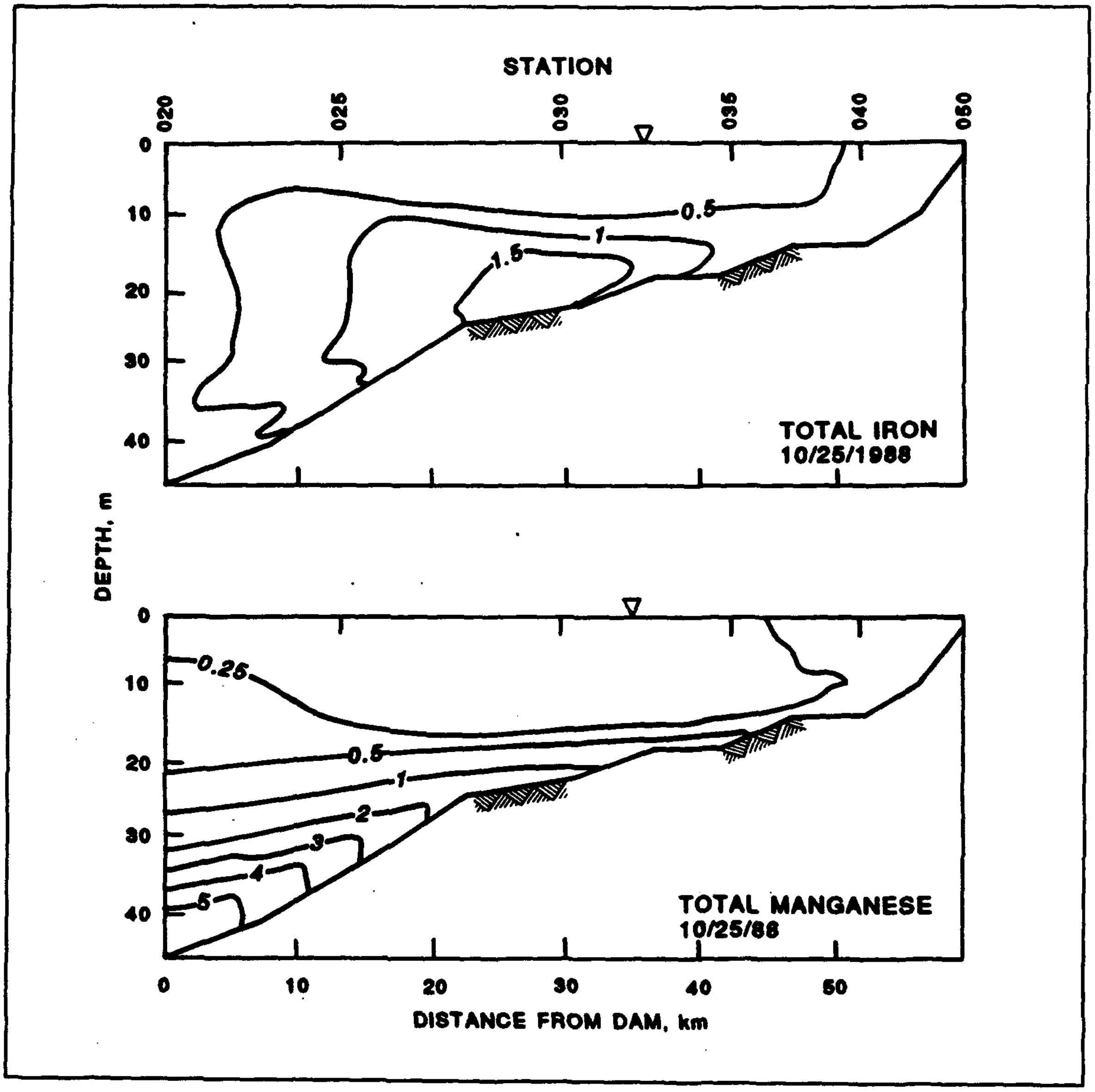

Figure 26. Spatial distribution of total iron and manganese concentrations (mg/ℓ) in the main stem of J. Strom Thurmond Lake, October 1988

manganese (2.3 mg/ℓ) and iron (2.5 mg/ℓ) were observed at sta 036 in the Broad River embayment.

Temperatures and dissolved oxygen concentrations in release waters from J. Strom Thurmond Dam (i.e. sta 010; Figure 28) displayed seasonal trends reflective of conditions in the J. Strom Thurmond Lake forebay. Temperatures gradually increased from 9 to 10 °C (February through March) to 18 to 20 °C (August through November). Dissolved oxygen concentrations remained near 8 to 11 mg/ℓ from January through May, declined to less than 3 mg/ℓ from August through October, and gradually returned to near 7 mg/ℓ during

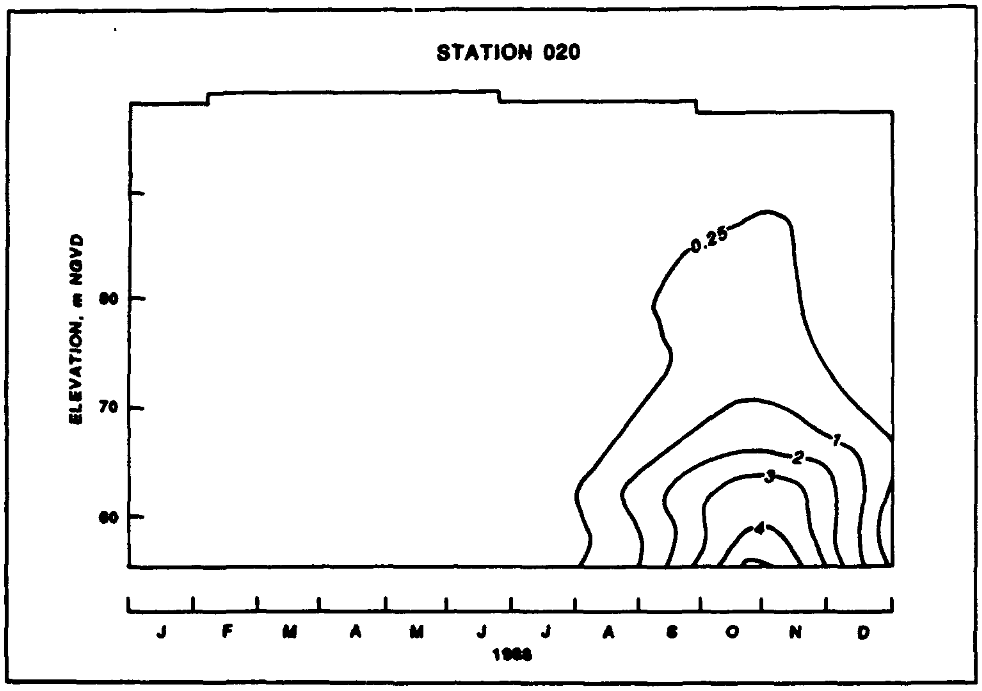

Figure 27. Temporal and vertical changes in dissolved manganese concentrations (mg/ℓ) in the forebay of J. Strom Thurmond Lake

October and November. By December, coincident with fall mixing in J. Strom Thurmond Lake, dissolved oxygen levels were near 10 mg/ℓ.

Chemical concentrations observed in release waters were reflective of conditions in the middepth waters of the J. Strom Thurmond Lake forebay. Maximum total manganese concentrations (0.3 mg/ℓ) were observed during October and dissolved manganese comprised the majority of the total manganese pool in the release waters. Maximum observed concentrations of total iron did not exceed 0.2 mg/ℓ during the study period. Nitrogen and phosphorus concentrations were near 0.4 mg/ℓ and 0.01 mg/ℓ, respectively. Values of conductivity, pH, alkalinity, organic carbon, and phosphorus were similar to values observed in the forebay.

Oxygenation System Operation

Operation of the oxygenation system was initiated on May 11, 1988 with a delivery rate of 15 tons[1] of oxygen per day through the pulse injection system located at the Richard B. Russell Dam face. A rate of 15 to 30 tons of oxygen per day was maintained through the pulse injection system until August 22, at

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[1] A table of factors for converting non-SI units of measurement to SI units is presented on page v.

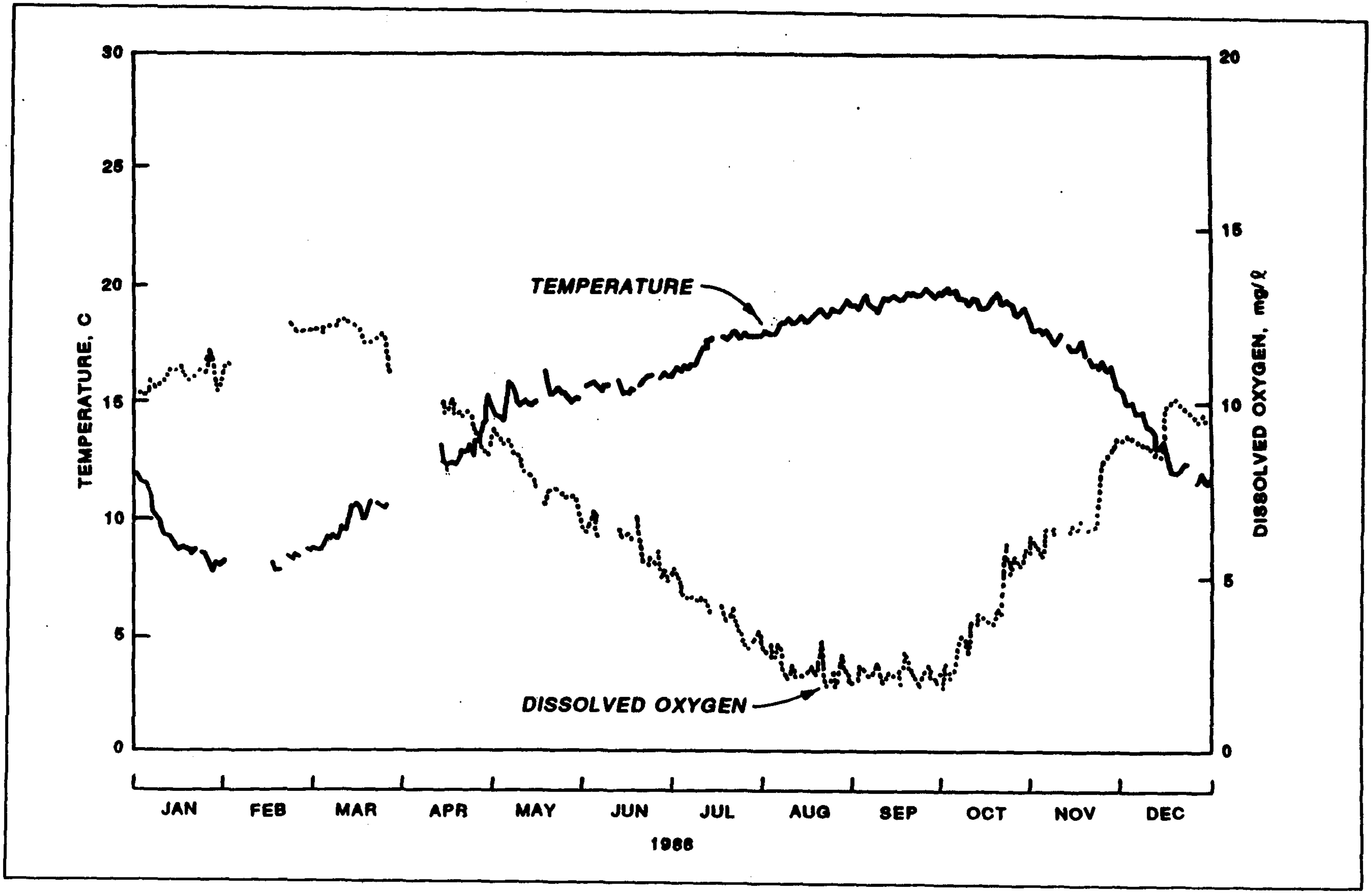

Figure 28. Temperatures and dissolved oxygen concentrations in the J. Strom Thurmond Dam releases, sta 010, 1988

which time, operation of the continuous injection system (located 1 mile upstream of the dam) was initiated. At this time, the continuous injection system delivered oxygen at a rate of 25 to 30 tons per day while the rate at the pulse injection system was 30 to 35 tons of oxygen per day. Both systems were operated throughout the remainder of the stratified period (until mid-October) at a combined capacity near 65 tons of oxygen per day. Delivery rates were decreased to between 30 and 50 tons of oxygen per day until early November and the system was turned off on November 9, 1988.

Operation of the oxygenation system maintained the concentration of dissolved oxygen near 6 mg/ℓ, at most depths, in the area affected by the system. Dissolved oxygen concentrations below 6 mg/ℓ were observed in the Richard B. Russell forebay (i.e. sta 060B) only at depths below 35 m with the exception of occasional concentrations near 4 mg/ℓ occurring in the metalimnion during September and October (see Figure 10). Additionally, concentrations of dissolved oxygen in the releases from Richard B. Russell Dam were maintained near 6 mg/ℓ with operation of the oxygenation system (Figure 29).

The water quality model SELECT (Davis et al. 1987) was employed to predict outflow oxygen concentrations based on lake data from sta 060B and 120 in the presence and absence, respectively, of oxygen system operation. These predictions and the actual outflow concentrations are shown in Figure 30. Results from SELECT indicate that the oxygenation system adds approximately 2-4 mg/ℓ dissolved oxygen to Richard B. Russell Dam releases.

Phytopigment Distributions

Seasonal and spatial trends in chlorophyll *a* concentrations in Richard B. Russell and J. Strom Thurmond Lakes were different from those observed in previous years of the study (Hains et al. 1988; Ashby et al. 1990). Whereas a single concentration peak was observed in each of the previous years, usually in late October, two concentration peaks were observed in 1988 (Figure 31). In January, concentrations were low (mostly <10 μg/ℓ) at all locations but minor spatial trends were apparent in each lake with concentrations generally increasing in a downstream direction. In April, concentrations increased to near 20 μg/ℓ in both lakes. Spatial trends were more pronounced in Thurmond Lake during this time. Concentrations decreased to near 5 μg/ℓ in July and were similar throughout both lakes. Concentrations again increased in October to approximately 10 μg/ℓ in Richard B. Russell Lake and to near 20 μg/ℓ in the midstream region of Thurmond Lake. Spatial trends within Thurmond Lake were again more pronounced than those in Richard B. Russell Lake. Distribution patterns in October were similar to those observed in April.

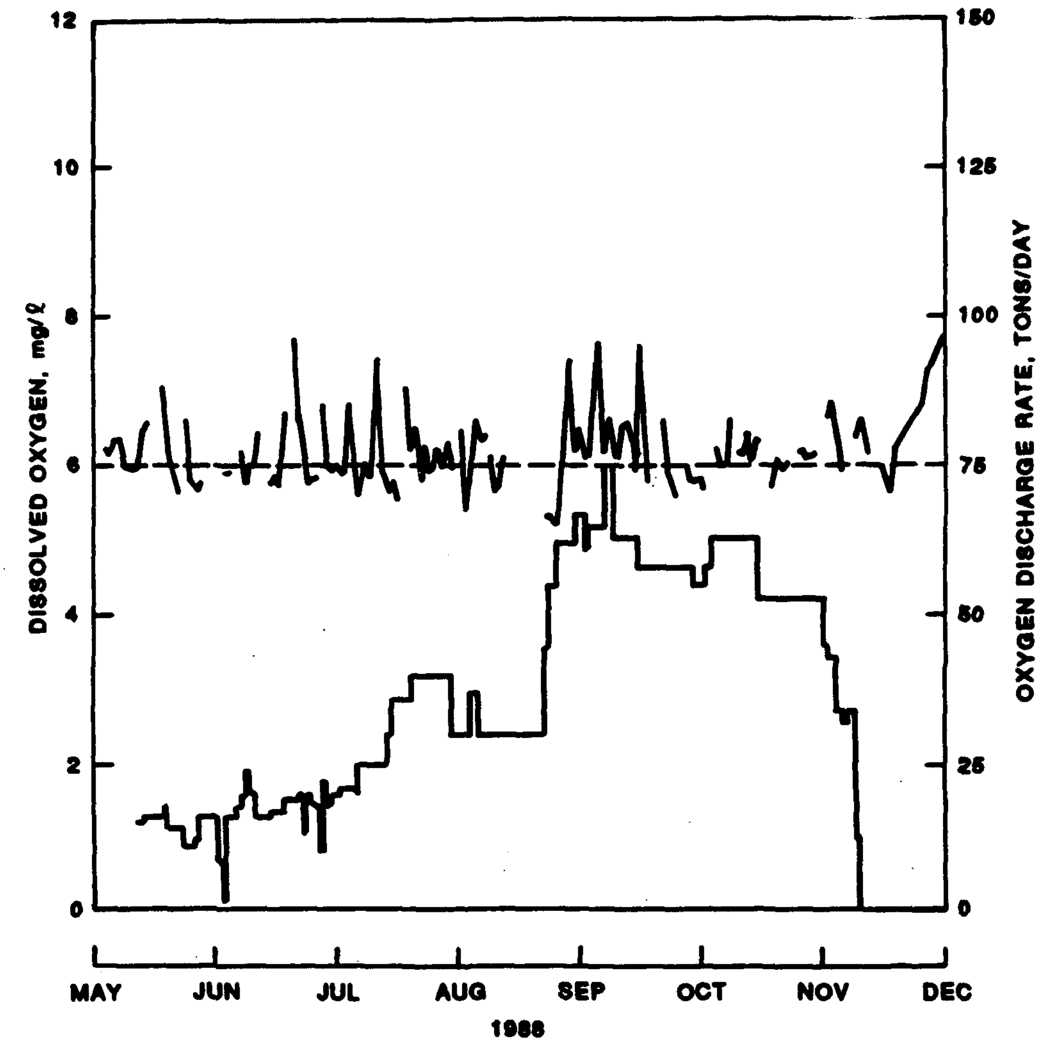

Figure 29. Dissolved oxygen concentrations (broken line) in the Richard B. Russell Dam releases and the oxygen delivery rate from the oxygenation system (solid line). Horizontal dashed line indicates the 6-mg/ℓ target dissolved oxygen concentration

Sediment Trap Study

A special study designed to describe the nature and rates of sediment deposition in the forebay area of Richard B. Russell Lake and the upstream regions of J. Strom Thurmond Lake was initiated during April 1988. The primary objective was to identify distribution patterns of sediment deposition; particular emphasis was on patterns of particulate metals related to oxygenation of the Richard B. Russell Lake hypolimnion.

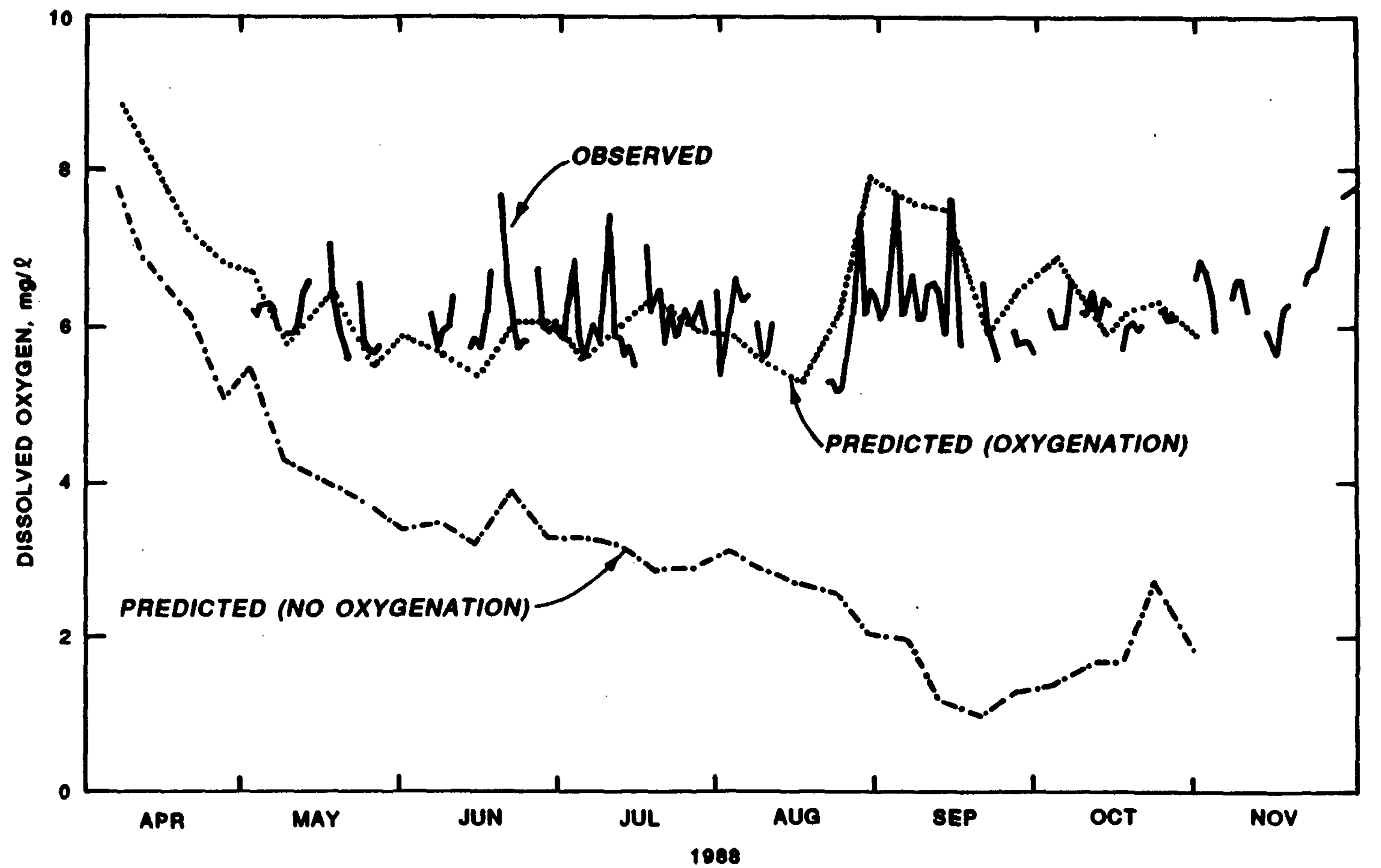

Figure 30. SELECT predictions of dissolved oxygen concentrations versus observed concentrations in the Richard B. Russell Dam releases.

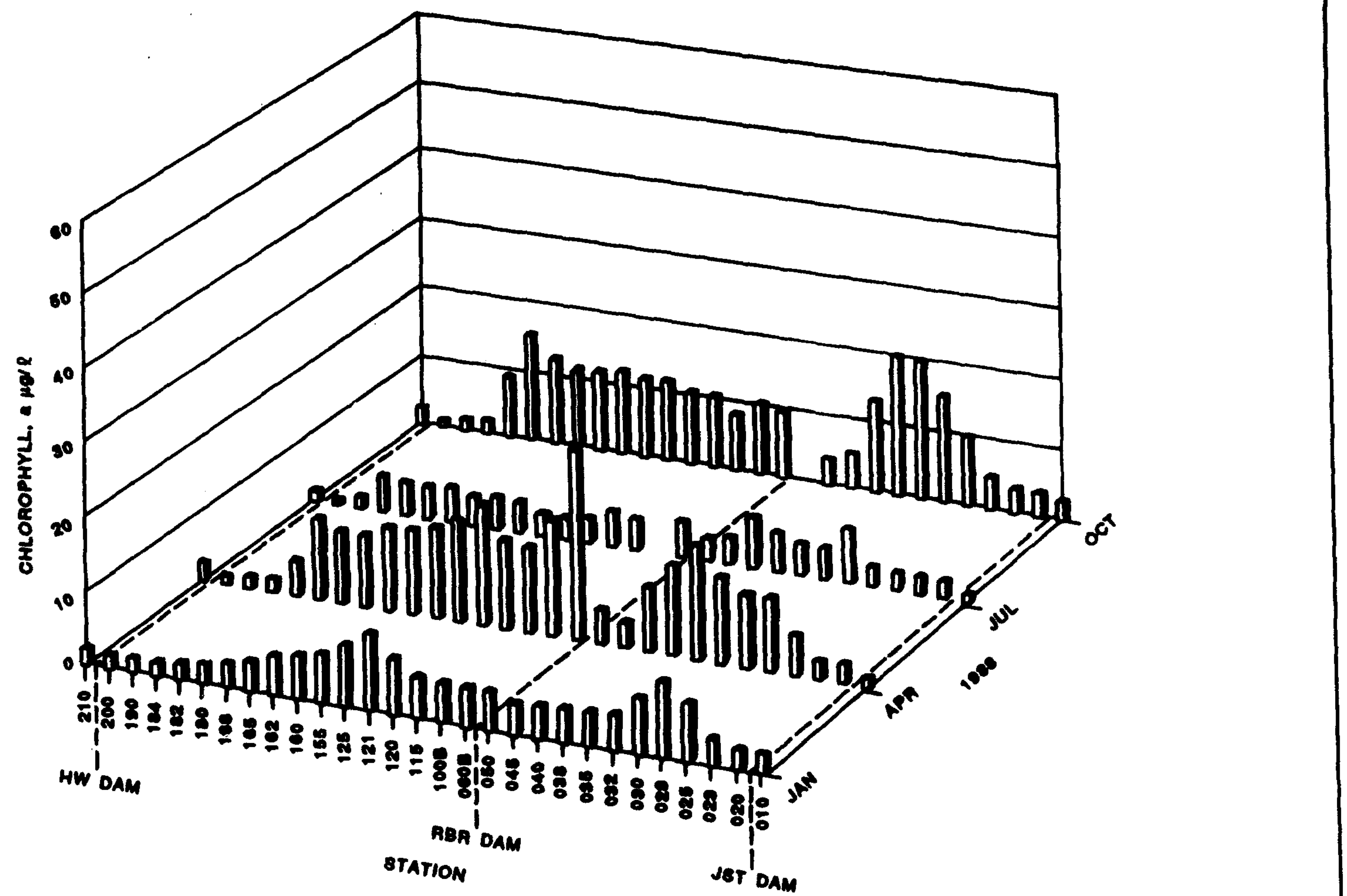

Figure 31. Patterns of distribution of chlorophyll *a* concentrations (μg/ℓ) in Hartwell, Richard B. Russell, and J. Strom Thurmond Lakes for January, April, July, and October 1988

Methods followed recommendations by Blomqvist and Håkanson (1981) and Bloesch and Burns (1980) so that properly designed sediment traps would allow accurate estimation of sediment deposition. Five locations (Figures 2 and 3), sta 120, 100B, 060B, 040, and 038, were chosen to represent respective regions of Richard B. Russell and J. Strom Thurmond Lakes. Stations 038 and 040 located below and above, respectively, the confluence of Broad River, GA, and below Richard B. Russell Dam are considered to be representative of the J. Strom Thurmond Lake headwater area. Stations 060B and 100B are located in the Richard B. Russell Lake forebay in the regions of the pulse and continuous oxygenation systems. Station 120 in Richard B. Russell Lake is considered to be out of the influence of hypolimnetic oxygenation and, therefore, provides a control for comparison among stations. Assemblies of six traps were suspended using an anchor and a submerged buoy. Approximately monthly deployments were employed to allow a complete record of temporal and spatial sediment deposition patterns. Triplicate tubes from each assembly and three replicates of one were analyzed for metals (iron and manganese), total nutrients (carbon, nitrogen, and phosphorus), dry mass, and ashed residue.

Maximum deposition rates for total dry mass in 1988 were observed primarily during summer months with a second peak occurring in winter, and deposition rates in J. Strom Thurmond Lake markedly exceeded rates observed in Richard B. Russell Lake (Figure 32). Total dry mass exceeded 200 g/sq m per day in the upstream regions of J. Strom Thurmond Lake during August and September and were near 200 g/sq m per day in December. Nonvolatile ash residue (Figure 33) did not reflect a bimodal distribution and pronounced rates of deposition of the inorganic ash residue occurred in the upstream region of J. Strom Thurmond Lake only during the winter. At that time J. Strom Thurmond Lake was 3 m below maximum conservation pool elevation and numerous deposits of old sediments immediately downstream from Richard B. Russell Dam were exposed or near the lake surface, potentially providing a source of inorganic material. Deposition of volatile or organic mass, the difference between total dry mass and nonvolatile ash residue, was similar to total dry mass rates and suggests the majority of deposited material is organic. The source of this organic material has not yet been determined.

Assessments of inorganic materials being deposited were focused on iron and manganese to delineate effects of oxygenation system operation on deposition patterns. Iron deposition (Figure 34) initially increased in J. Strom Thurmond Lake receiving waters during July and peaked during August (greater than 15 g Fe/sq m per day) while the pulse oxygenation system was in operation. A dramatic decrease in iron deposition in J. Strom Thurmond Lake was observed in September following the addition of the continuous oxygenation system in Richard B. Russell Lake (Figure 35). Furthermore, increased iron deposition rates at sta 060B in Richard B. Russell Lake were observed after operation of the continuous oxygenation system began. The coincidental decline of deposition in J. Strom Thurmond Lake in September may be due to trapping of particulate iron in Richard B. Russell Lake in response to operation of the continuous oxygenation system and increased hydraulic residence in

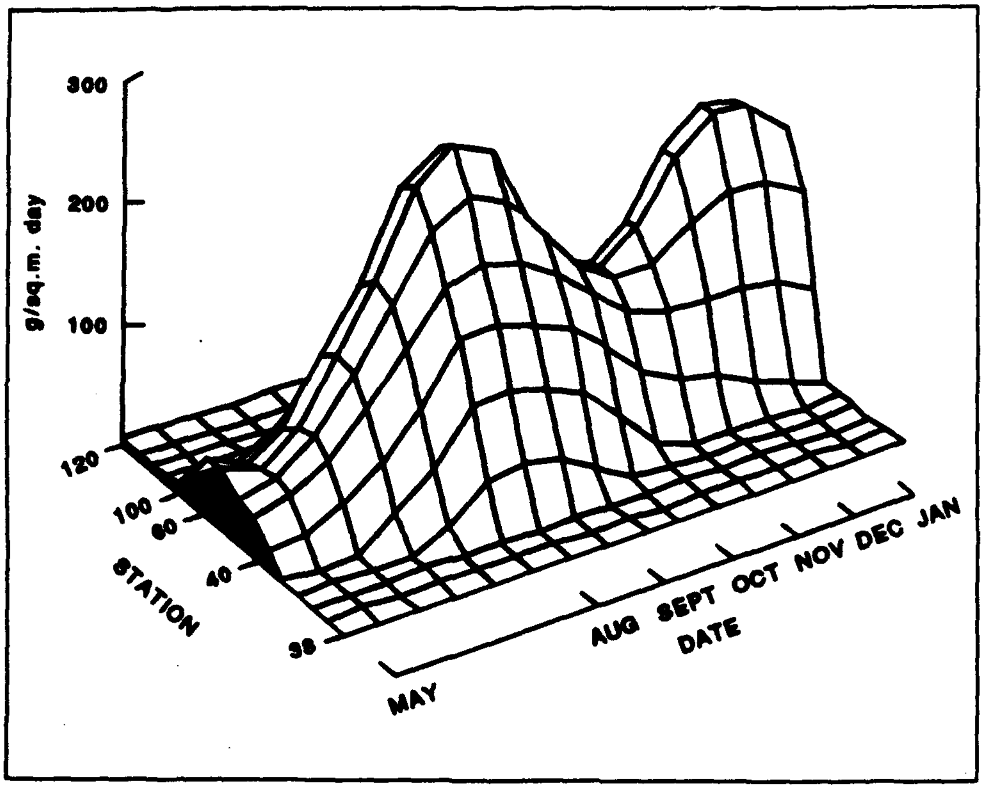

Figure 32. Patterns of dry weight deposition rates in Richard B. Russell and J. Strom Thurmond Lakes, May 1988 through January 1989

Richard B. Russell Lake from sta 100B to the dam. However, maximum iron deposition rates in Richard B. Russell Lake were less than 50% of the J. Strom Thurmond Lake maximum rate.

Deposition of manganese (Figure 36), like iron, was greater downstream of Richard B. Russell Dam; however, temporal deposition occurred later at sta 040 and 038 than that of iron. Maximum rates of manganese deposition (nearly 3 g Mn/sq m per day) were observed at sta 040 in September. Again this followed initiation of operation of the continuous oxygenation system (Figure 37). And again, the decline of manganese deposition in J. Strom Thurmond Lake accompanied increases in Richard B. Russell Lake in October (sta 060B) and December (sta 100B and 120). In Richard B. Russell Lake, increased deposition of manganese at sta 100B and 120 is probably in response to winter mixing upstream. This could allow sufficient residence time for manganese oxidation and subsequent deposition as particulates.

Deposition rates of phosphorus were similar to those observed for total dry mass and were representative of deposition rates of nitrogen and organic mass. Patterns of deposition rates typified temporal and spatial observations previously described for total dry mass.

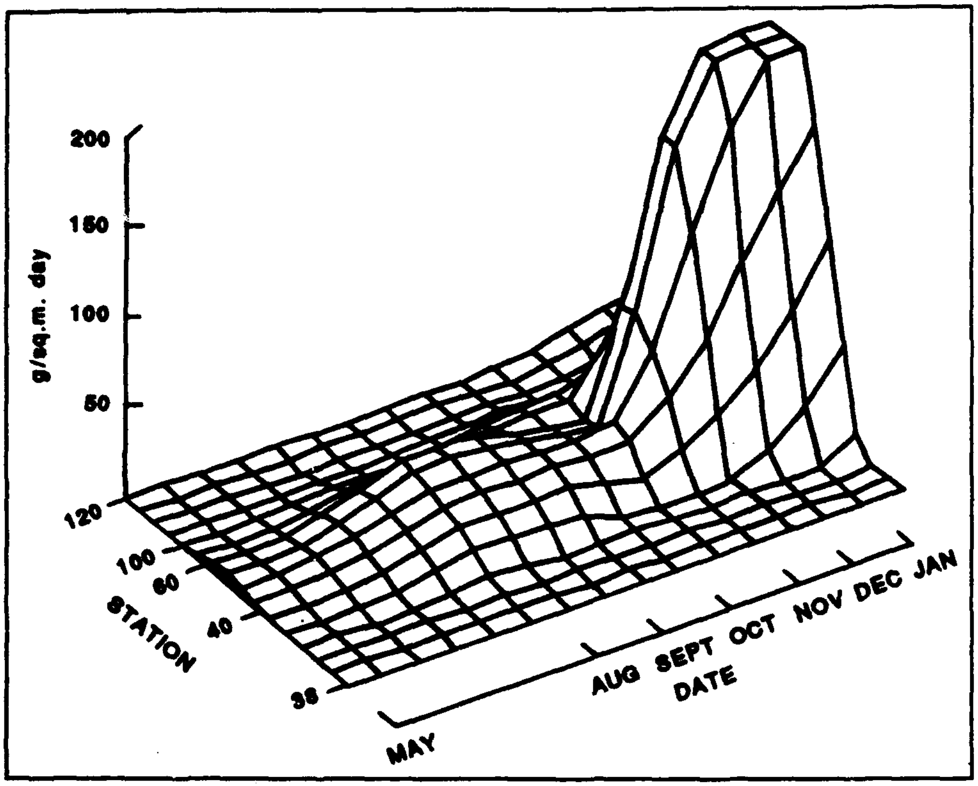

Figure 33. Patterns of ash weight deposition rates in Richard B. Russell and J. Strom Thurmond Lakes, May 1988 through January 1989

Patterns of deposition downstream from Richard B. Russell Lake are complex. Dry mass deposition showed a clear bimodal distribution at sta 040. The first peak occurred early in the study during times of greater flow and influence by Broad River, GA. This inflow did influence the areas near sta 040 during times of nongeneration and the decline of the first peak accompanied the onset of a low rainfall period, reduced inflows, and sediment loads. Lake elevations in J. Strom Thurmond Lake declined during this time also, although Richard B. Russell Dam continued scheduled releases. Comparison of distribution of dry mass deposition with ash residue deposition shows that dry mass of the second peak in deposition was of different composition than composition of dry mass of the first peak. At the time of the later peak of dry mass deposition, deposition of metals in J. Strom Thurmond Lake were decreased, suggesting that inorganic materials other than metals were probably responsible for the later peak. One possibility is that minimal lake levels which exposed old sediment deposits to outflows from Richard B. Russell Dam contributed to the redeposition of these sediments downstream in the sediment traps. Such a response is feasible and would explain the inorganic nature of deposited materials and the relative lower metals content.

The study has thus far shown that particulate metals are being deposited in Richard B. Russell and J. Strom Thurmond Lakes in response to lake

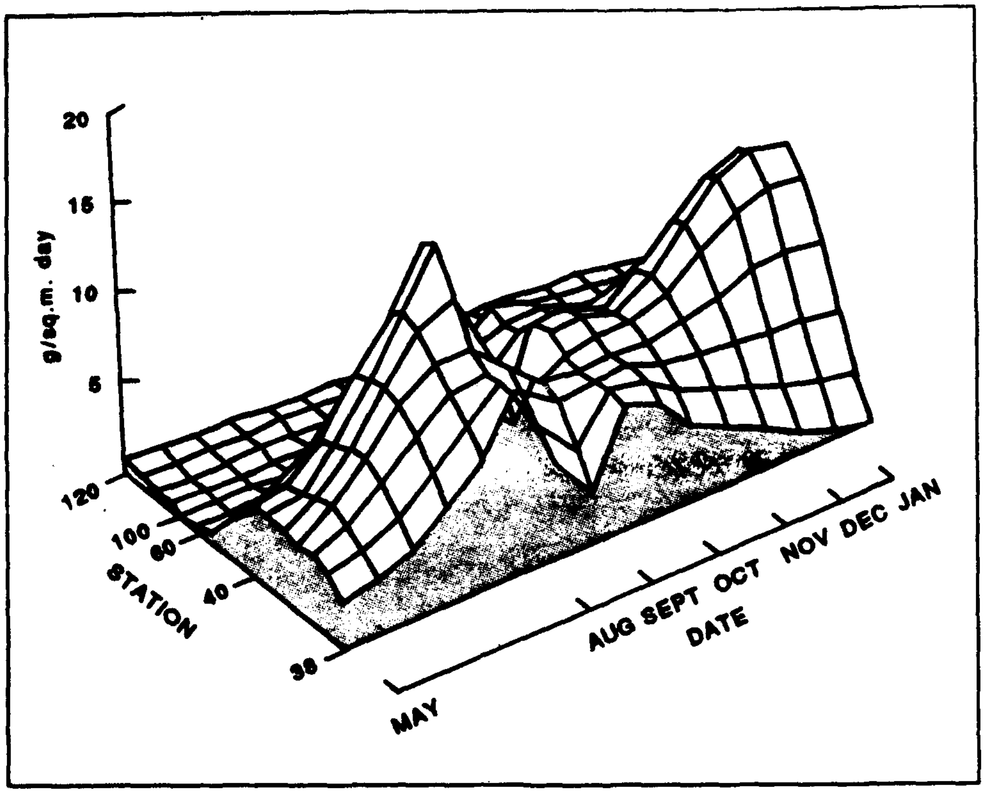

Figure 34. Patterns of iron deposition rates in Richard B. Russell and J. Strom Thurmond Lakes, May 1988 through January 1989

oxygenation and that operation of different oxygenation systems can influence the patterns of deposition. Materials deposition in J. Strom Thurmond Lake is greater than that for the downstream region of Richard B. Russell Lake. Furthermore, deposition of materials in J. Strom Thurmond Lake is influenced by both tributary inflows (Broad River, GA) and lake surface elevations as well as the oxygenation systems in Richard B. Russell Lake.

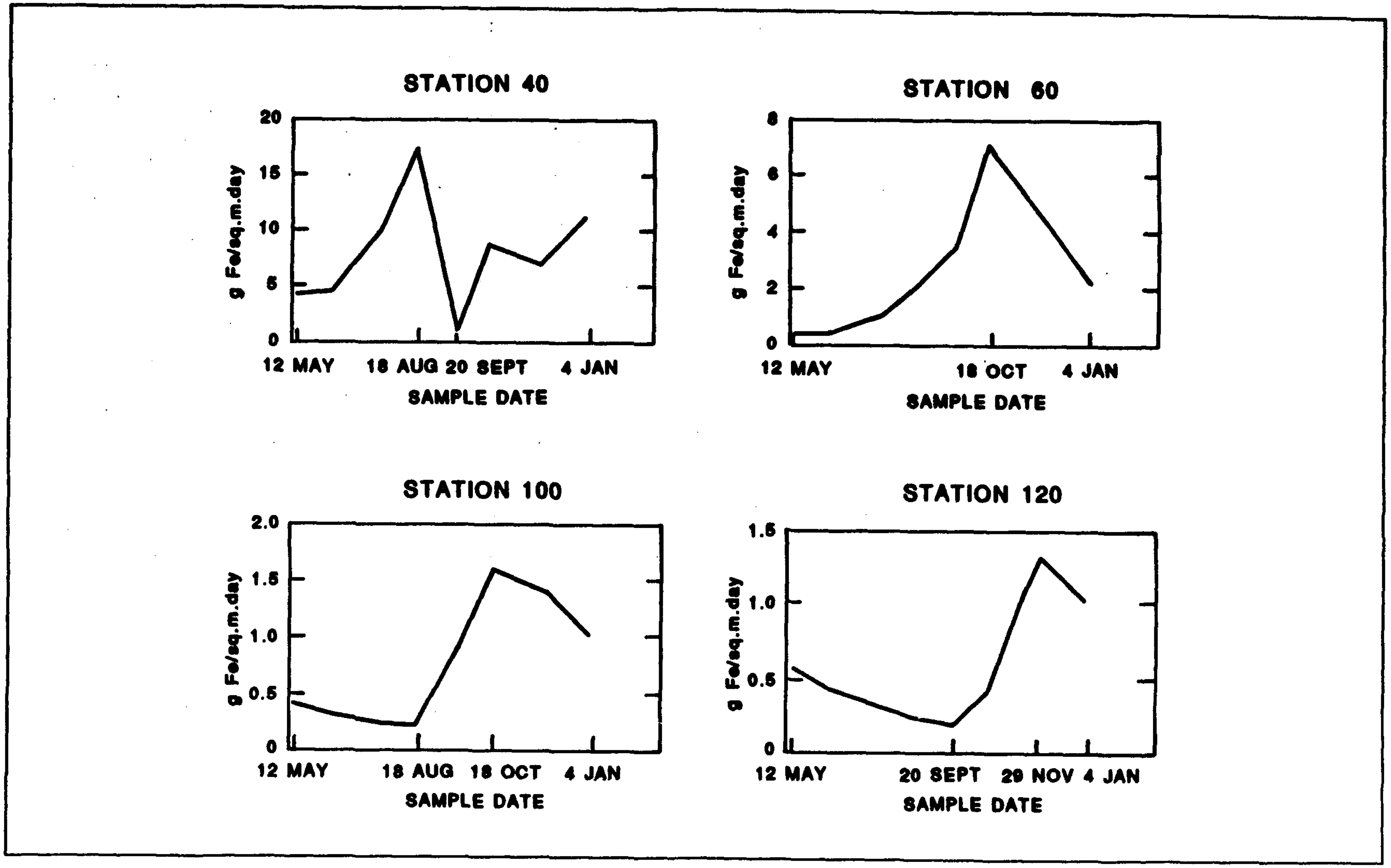

Figure 35. Patterns of iron deposition rates in Richard B. Russell and J. Strom Thurmond Lakes, May 1988 through January 1989

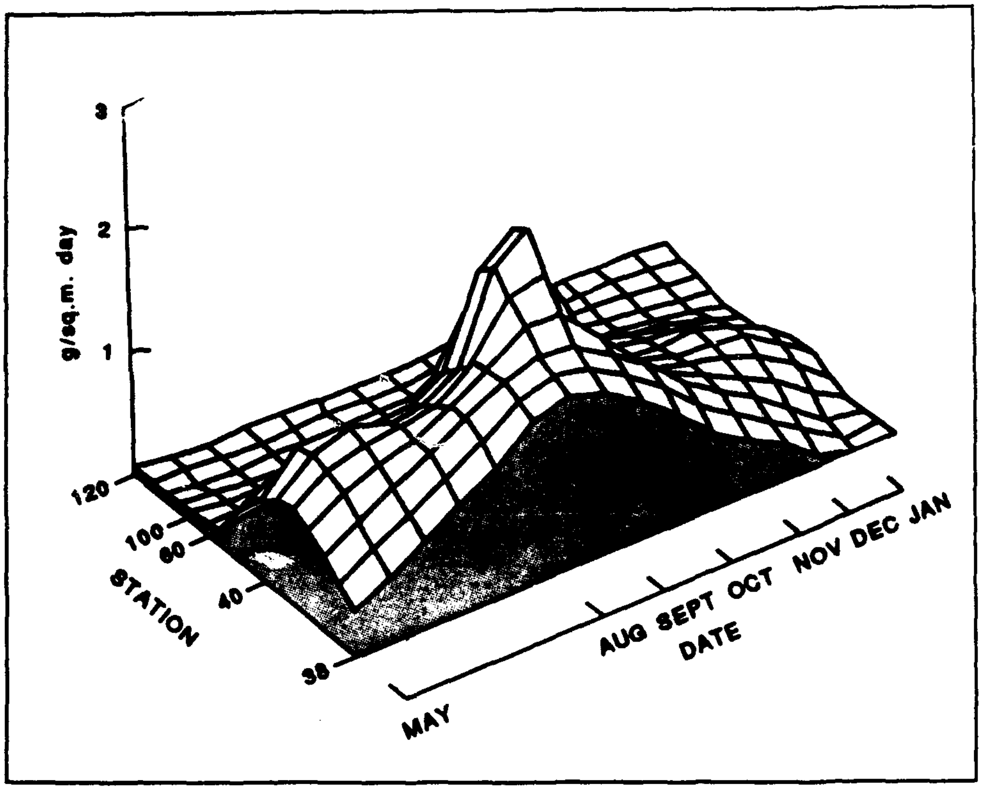

Figure 36. Patterns of manganese deposition rates in Richard B. Russell and J. Strom Thurmond Lakes, May 1988 through January 1989

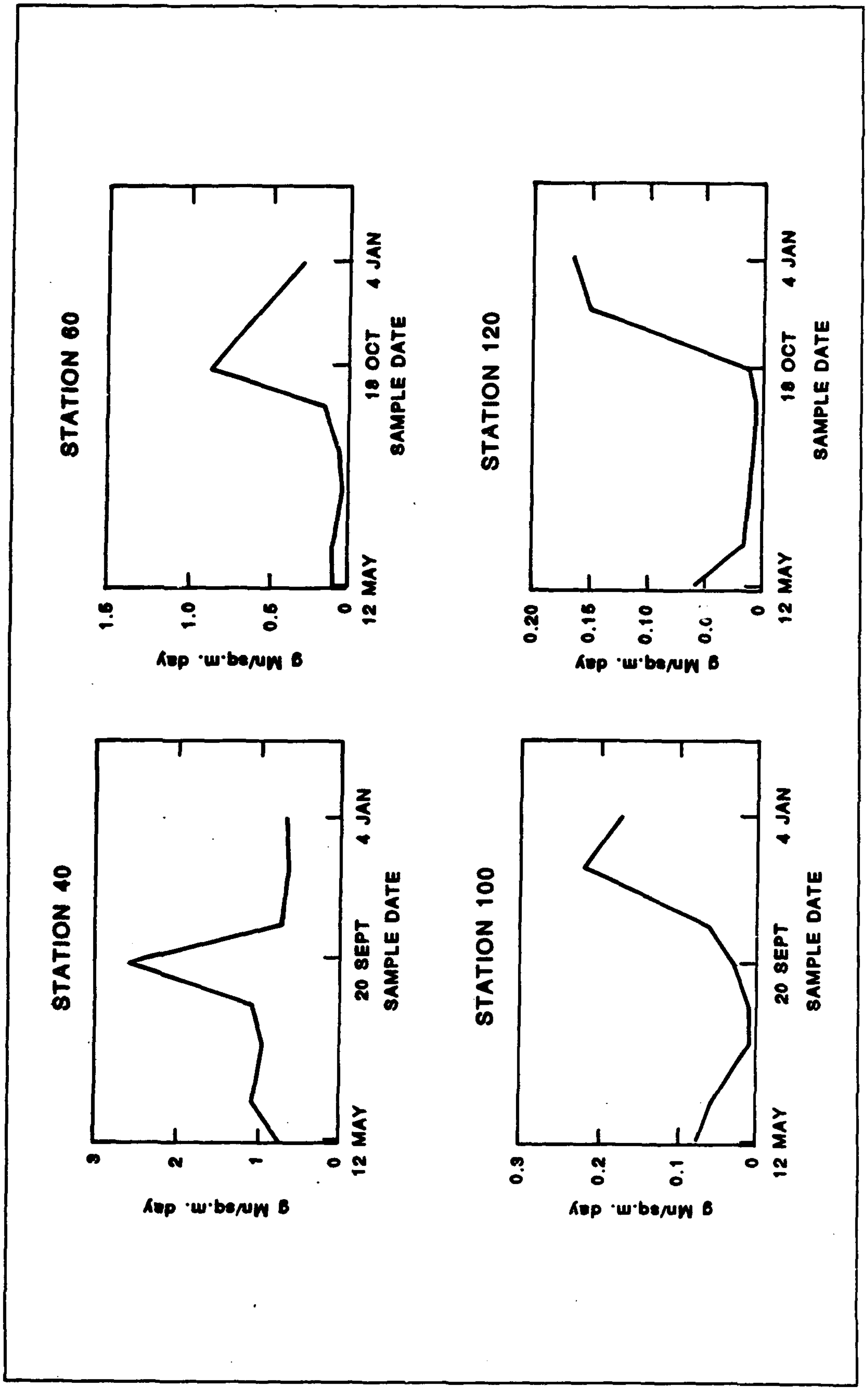

Figure 37. Patterns of manganese deposition rates in Richard B. Russell and J. Strom Thurmond Lakes, May 1988 through January 1989

5 Discussion

Water Quality Conditions During 1988

Temporal and spatial trends in thermal structure observed in 1988 in Hartwell, Richard B. Russell, and J. Strom Thurmond Lakes were similar to those observed in previous studies (Ashby et al. 1990; Hains et al. 1988; James et al. 1986, 1985). The onset of stratification occurred in March or April in each lake, and a well-developed thermocline was established by May. Thermal stratification intensified during the summer due to increased warming of surface waters. As daily air temperatures began to decrease in late September, surface water temperatures decreased and vertical gradients in water density diminished. Consequently, seasonal mixing or destratification occurred during October and November and isothermal conditions existed throughout the winter period.

As the hypolimnia of each lake became isolated during thermal stratification, oxygen depletion occurred in bottom waters. Oxygen depletion and development of anoxic conditions were more pronounced in areas with limited aeration and increased levels of organic matter. These areas were typically located in the deep, forebay regions of the main stems and tributary embayments of the lakes. In contrast, bottom waters in the mid- and upstream regions in the main stems of Richard B. Russell and J. Strom Thurmond Lakes remained well-oxygenated during thermal stratification. In Richard B. Russell Lake this may be attributed to well-oxygenated releases from Hartwell Lake for most of the stratified period and natural aeration in the Richard B. Russell Lake headwater region. Release of oxygenated water from Richard B. Russell Dam contributed to well-oxygenated waters in the mid- and upstream regions of J. Strom Thurmond Lake. Increased dissolved oxygen concentrations, observed after destratification in bottom waters not influenced by the oxygenation system, can be attributed to wind-induced mixing and natural aeration.

Among-lake variations in the development of anoxic conditions and reaeration associated with seasonal mixing were most pronounced in bottom waters of each lake. While the tributary embayments of both J. Strom Thurmond and Richard B. Russell Lakes developed anoxic conditions near early June, anoxic conditions in the downstream regions in each lake and in Hartwell Lake developed at different times during the stratified period. Anoxic conditions

developed in the downstream region of Richard B. Russell Lake (i.e. sta 120) by June but were not well established in the forebay region of J. Strom Thurmond Lake until late August; anoxic conditions were not present in Hartwell Lake until mid-October. Additionally, oxygen concentrations associated with seasonal mixing varied in Hartwell Lake as well. Dissolved oxygen concentrations in Richard B. Russell and J. Strom Thurmond Lakes increased to near 8 mg/ℓ in late October; however, concentration increases were not apparent in bottom waters in Hartwell Lake until late November and December. Among-lake variations may be the result of differences in oxygen regimes (processes of utilization and production of dissolved oxygen) in each lake and a temporal function of seasonal mixing.

Varied thermal structure and resultant dissolved oxygen regime in the downstream region of the lakes may be attributed to morphometric, inflow, or operational characteristics. A more pronounced thermal structure in Hartwell Lake (i.e. deeper thermocline depth and cooler mean hypolimnetic temperature) contributes to approximately one to two months lag in development of hypolimnetic anoxia and fall mixing in the Hartwell Lake forebay (Ashby et al. 1990). In Richard B. Russell and J. Strom Thurmond Lakes, temperature and dissolved oxygen regimes in the main stem are influenced by inflows from upstream hydropower projects (Hartwell and Richard B. Russell Lakes, respectively). Typically, these inflows are cool and well-oxygenated and effectively maintain favorable conditions in the upstream region of the main stems. The proximity of major secondary tributaries to the mid- and downstream regions of Richard B. Russell and J. Strom Thurmond Lakes may contribute to development of anoxia in downstream region hypolimnia via increased material loads.

Temporal changes in water quality were more pronounced in major embayments than at main stem locations due presumably to morphology and loading. Higher concentrations of organic carbon and reduced metals in the embayments may contribute to increased oxygen depletion and intensify anoxic processes, as suggested by higher oxygen depletion rates at embayment stations than at main stem locations. Morphometric characteristics and increased hydraulic retention times at embayment stations may result in higher material load per unit volume. Detailed loading data are not available for robust comparison, however.

Operation of the oxygenation system in the hypolimnion of the downstream region of Richard B. Russell Lake markedly influences water quality in the vicinity of the system, in release waters, and in J. Strom Thurmond Lake. Operation of the oxygenation system maintained concentrations of dissolved oxygen near 6 mg/ℓ in the forebay and releases from Richard B. Russell Dam during stratification. Without operation of the oxygenation system, outflow concentrations (based on predictions from SELECT) would have been approximately 2 mg/ℓ lower, potentially impacting downstream water quality and biotic communities.

Additional effects of the oxygenation system include effects on the dynamics of iron and manganese in the area influenced by the system. Distribution patterns of dissolved and particulate iron and manganese concentrations observed during 1988 were similar to those of previous years (Ashby et al. 1990, Hains et al. 1988, James et al. 1986). Oxidation and resultant precipitation of reduced iron is enhanced in the presence of the oxygenation system, as indicated by increased concentrations of particulate iron in the area affected by the system. Concentrations of particulate manganese were mostly near detection limits downstream of the system, indicating that the effects of the oxygenation system on the oxidation of reduced manganese are minimal.

Oxygen depletion in sediments and overlying water results in solubilization and increased mobility of metals and the establishment of concentration gradients via advection and diffusion. Vertical gradients in manganese and iron concentrations established as anoxic conditions developed in the hypolimnia of tributary embayments and in bottom waters in the downstream regions of the lakes. However, concentrations of manganese and iron were near the detection limit except during thermal stratification and periods of high flow. Manganese was observed in measurable quantities primarily in dissolved or reduced forms coincident with anoxia in each lake. Conversely, iron was primarily in particulate or oxidized forms except in anoxic regions.

Among-lake variations in manganese and iron concentrations were discernible in the downstream regions of the main stems. Dissolved manganese concentration maxima were higher in the J. Strom Thurmond Lake main stem (> 5 mg/ℓ) than in the main stems of Richard B. Russell and Hartwell Lakes (< 2 mg/ℓ in each lake). Conversely, highest iron concentrations were observed in the downstream region of Richard B. Russell Lake (> 6 mg/ℓ), while main stem concentration maxima in Hartwell and J. Strom Thurmond Lakes were less than 2 mg/ℓ in each lake. Mechanisms for these distribution variations were not delineated, but may partially be the result of varied intensity of anoxic processes in each lake and temporal variability not adequately described with current sampling intervals. Maintenance of well-oxygenated hypolimnetic waters in the Richard B. Russell Lake forebay inhibits reduction and solubilization of manganese from the sediments and could account for observations of lower concentrations. Similarly, Hartwell Lake exhibits anoxia in the forebay very briefly and late in the stratified period, resulting in a short period conducive to reduction and solubilization of manganese from the sediments. Water quality sampling not coincident with this period would not reflect increased concentrations. Iron, however, requires a lower oxidation-reduction potential for reduction and solubilization but is more readily oxidized than manganese. Influences of the oxygenation system on the oxidation of iron may account for observations of concentration maxima in Richard B. Russell Lake.

Seasonal reduction, solubilization, and transport of iron and manganese coupled with oxidation, deposition, and redistribution result in complex interactions in iron and manganese cycling in the three-lake system. These interactions may be attributed to inflow mixing patterns, operation of the oxygenation

system in Richard B. Russell Lake, and fluctuations in lake elevation at J. Strom Thurmond Lake. For instance, inflow mixing patterns contribute to deposition patterns of influent material (Håkanson and Jansson 1983) and are most important during high inflow events (Kennedy et al. 1980). Additionally, increased concentrations of dissolved materials may be present in tributary embayments and hypolimnia during anoxia, thereby increasing availability for advective transport. As a consequence, deposition patterns are further effected via oxidation of reduced materials and formation of particulates.

Sediment trap data indicate materials deposition in the downstream region of Richard B. Russell Lake and headwater region of J. Strom Thurmond Lake and may be attributed to operation of the oxygenation system, hydrologic and hydraulic influences, and seasonal variations in water quality. Increased iron deposition in Richard B. Russell Lake in conjunction with operation of the continuous oxygen injection system and increased iron deposition in J. Strom Thurmond Lake with operation of the pulse injection system delineate operational effects of hypolimnetic oxygenation on iron deposition. Clearly, rapid oxidation and resultant precipitation of reduced iron in the area affected by the oxygenation systems result in observed deposition patterns. Maximum deposition rates at sta 040 indicate that hydraulic interactions in the headwater region of J. Strom Thurmond Lake dominate distribution of materials. Energy regimes near sta 040 are greatly reduced from upstream, headwater regions and could account for observed increased deposition rates. Furthermore, fluctuating lake elevation at J. Strom Thurmond Lake may influence deposition patterns as well. Lake elevations below normal, experienced during periods of low inflow, expose sediments to hydrologic transport mechanisms, such as wind/wave erosion processes, that may not be as pronounced at times of higher lake elevations. Consequently, sediments not strongly subjected to transport at maximum conservation elevation may be redistributed during periods of lower elevations (i.e. increased transport due to scour effects). Lastly, seasonal variation in composition of sediment trap material suggests qualitative and quantitative responses of materials deposition to hydrologic and hydraulic characteristics. Changes in sediment composition would be profoundly influenced by concentrations of water quality constituents which vary seasonally, such as iron and manganese. Increased materials transport during periods of elevated flow would result in seasonal variability of materials deposition as well.

Although nutrient cycling is enhanced during anoxia (Mortimer 1941, 1942), seasonal concentration gradients of nitrogen and phosphorus were not as pronounced as concentration gradients of iron and manganese. Total nitrogen concentrations remained relatively constant (0.5 to 1.0 mg/ℓ); however, concentrations of dissolved forms varied seasonally. Maximum nitrogen concentrations, observed in spring following the high flow period and in summer prior to peak phytoplankton productivity, were primarily dissolved forms and readily available for uptake by phytoplankton. Increased phytoplankton productivity (as suggested by increases in chlorophyll *a* concentrations) may have contributed to midsummer decreases of dissolved nitrogen concentrations in the surface waters. Oxidized forms of nitrogen, primarily nitrate, were

predominant in bottom waters prior to the onset of anoxic conditions. As the hypolimnia of Hartwell and J. Strom Thurmond Lakes became anoxic, nitrate concentrations decreased to near detection limit, primarily due to denitrification by faculative anaerobic microbes. Coincident with decreased bacterial nitrification in the anoxic hypolimnia, concentrations of ammonia increased. Conversely, oxidized forms of nitrogen comprised the majority of the total nitrogen in the bottom waters of Richard B. Russell Lake, downstream of the oxygenation system, at all times. Oxidized forms of nitrogen, predominant in the area influenced by oxygenation, may be the result of maintenance of well-oxygenated conditions, which favors biochemical reactions resulting in oxidized forms of nitrogen as end products. Following fall mixing oxidized forms of dissolved nitrogen again comprised the majority of the total nitrogen pool.

Unlike nitrogen, phosphorus concentrations did not increase appreciably during anoxia or high flow periods and, consequently, pronounced gradients were not observed. In fact, total phosphorus concentrations, mostly near 0.01 and 0.02 mg-P/ℓ, reflect values typical of oligotrophic to meso-eutrophic aquatic systems (Vollenweider 1968), suggesting potential limits on phytoplankton productivity in the three-lake system. However, detailed analyses of nutrient limitation and phytoplankton productivity have not been conducted and stringent conclusions are inappropriate.

Although chlorophyll *a* concentrations were mostly lower than 20 ug/ℓ in each lake, concentration peaks observed in April and October suggest development of seasonal patterns previously not observed (Ashby et al. 1990, Hains et al. 1988, James et al. 1986). Seasonal succession of phytoplankton species may result in temporal peaks in chlorophyll *a* concentrations (Wetzel 1983); however, delineation of sources for observed peaks is not possible without phytoplankton species identification.

Long-Term Trends in Water Quality

Temperatures observed in the forebay (sta 020) and midstream region (sta 030) of J. Strom Thurmond Lake during the 5-year study are depicted in Figure 38. Statistical comparison of mean hypolimnetic temperatures, calculated for the stratification period (May through October) for each year at depths greater than 12 m at sta 20, indicates cooler hypolimnetic temperatures (approximately 1 to 2 °C) since 1984 (Table 4). Additionally, thermal stratification appears to be more defined (i.e. sharper thermal gradients) and may be the result of decreased hypolimnetic temperatures. For example, in 1984 at sta 020 temperature differences in the thermocline region were approximately 4 °C while differences of 6 to 8 °C were observed in 1986 and 1988 (Figure 38) even though similar surface temperatures were observed in each of the 5 years.

Changes in dissolved oxygen regimes at Richard B. Russell and J. Strom Thurmond Lakes are apparent for recent years. While comparisons between

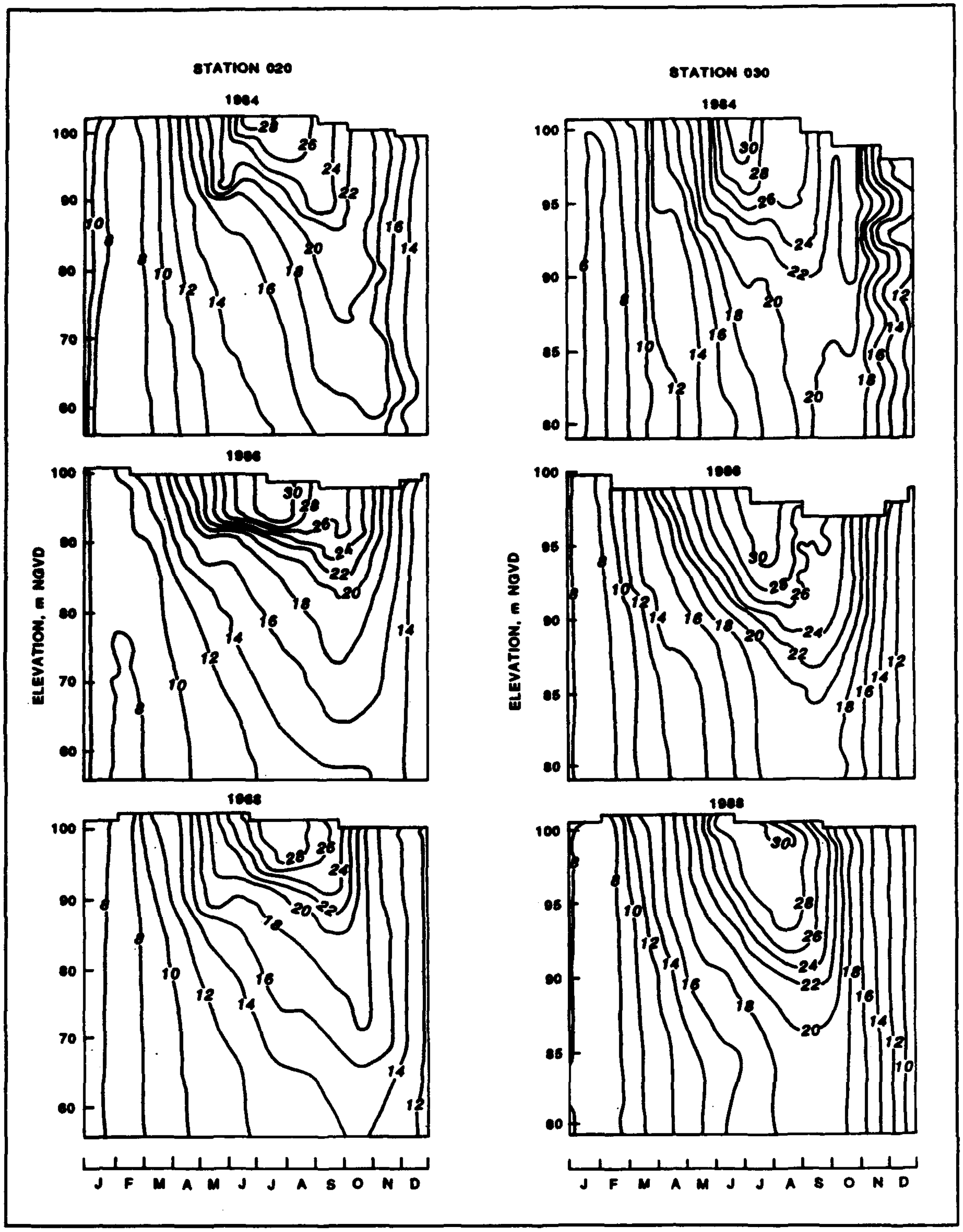

Figure 38. Temporal and vertical patterns in temperature (°C) for sta 020 and 030, J. Strom Thurmond Lake, 1984, 1986, and 1988

Table 4
Mean Hypolimnetic Temperatures in the Forebay and Midstream Region of J. Strom Thurmond Lake 1984-1988[1]

Year	Mean	n	Duncan Grouping[2]
1983	19.0	13	A
1984	16.8	144	B
1985	13.9	110	D
1986	15.3	101	C
1987	14.5	94	C/D
1988	14.9	83	C/D

[1] Calculated for the stratified period (May through October) at depth > 12 m.
[2] Means with the same letter are not significantly different at $\alpha = 0.05$.

lakes are difficult due to operation of the oxygenation system in Richard B. Russell, annual comparisons within each lake can be made. Most obvious in this comparison is a decrease in the extent of the anoxic zone in the forebay and downstream region of Richard B. Russell Lake since 1984 (Figure 39). Clearly, operation of the oxygenation system in the Richard B. Russell Lake forebay has resulted in the pronounced decrease of the anoxic region in the forebay region. However, decreased oxygen consumption of inundated materials is evidenced by marked decreases in anoxic regions in upstream regions which are not influenced by the oxygenation system (i.e. sta 120). Additionally, prior to 1987, anoxic conditions had been observed as far upstream as sta 160 (Hains et al. 1988, James et al. 1986, 1985). Conversely, dissolved oxygen conditions in the tributary embayments of Richard B. Russell have remained similar during the post-impoundment period (Figure 40). However, a slight decrease in the extent of the anoxic zone since 1985 indicates gradual, continued improvement in the oxygen regime in the tributary embayments. This slower rate of improvement may be attributed to decreased flushing and higher organic loads in the tributary embayments.

Dissolved oxygen conditions in the main stem of J. Strom Thurmond Lake have changed from 1984 to 1988 (Figure 41). The extent of the anoxic zone in the hypolimnion of the forebay region (sta 020) has increased. A number of factors may contribute to increases in the extent of the anoxic region, such as increased thermal structure, changes in oxygen-consuming materials loads, and effects of sustained decreased inflows during drought conditions. Limited loading data and extremes in hydrologic conditions prevent clear delineation of contributing factors, however. Conversely, while anoxic conditions were observed in the hypolimnion at sta 030 in 1984, dissolved oxygen levels have remained above 2 mg/ℓ since. Time-of-travel studies indicate that well-oxygenated release water from Richard B. Russell Dam markedly influences water quality in the mid- and upstream region of J. Strom Thurmond Lake (Ashby et al. 1990). Consequently, decreases in the extent of anoxic regions in the midstream region may be related to operation of the oxygenation system

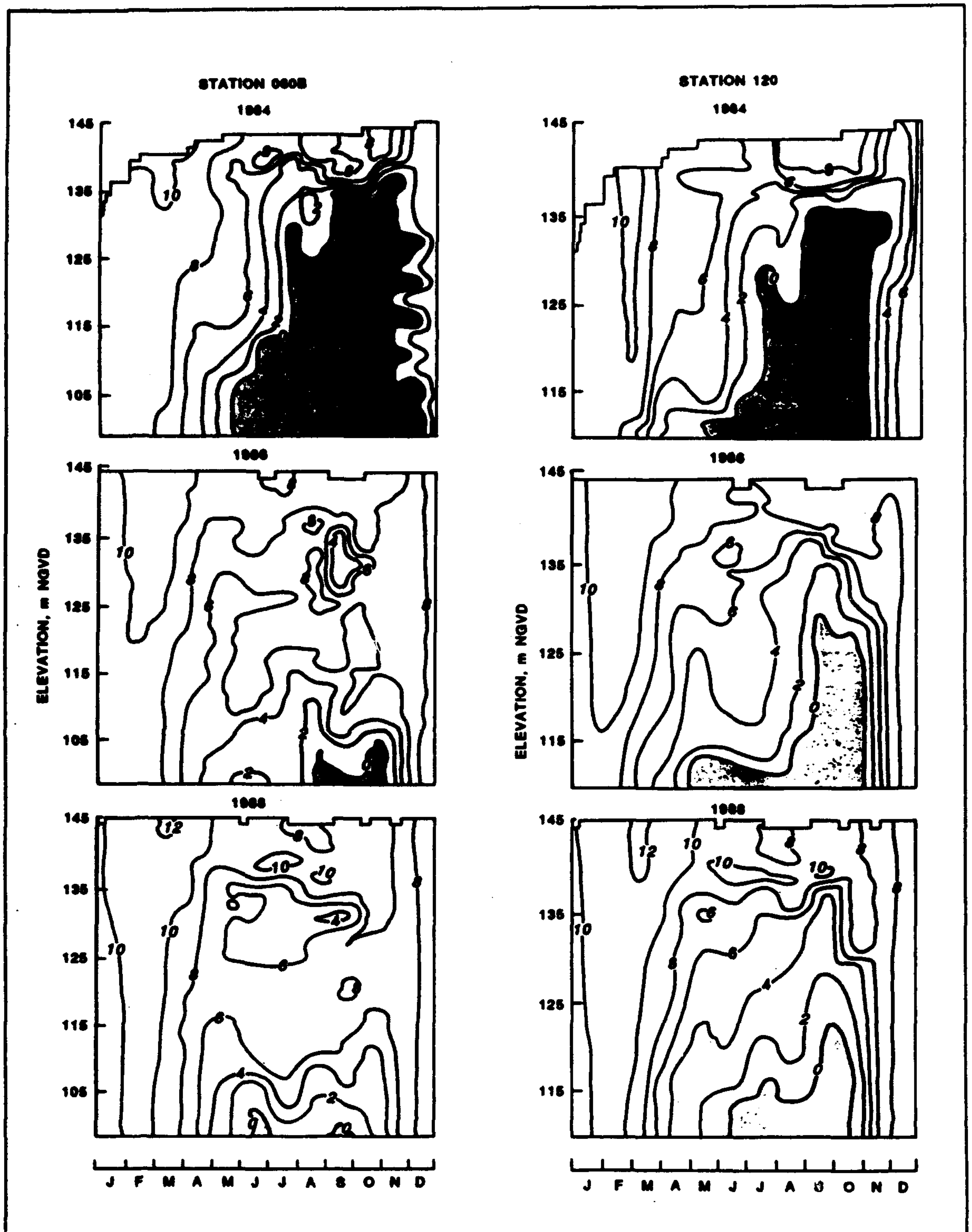

Figure 39. Temporal and vertical patterns in dissolved oxygen (mg/ℓ) for sta 060B and 120, Richard B. Russell Lake, 1984, 1986, and 1988

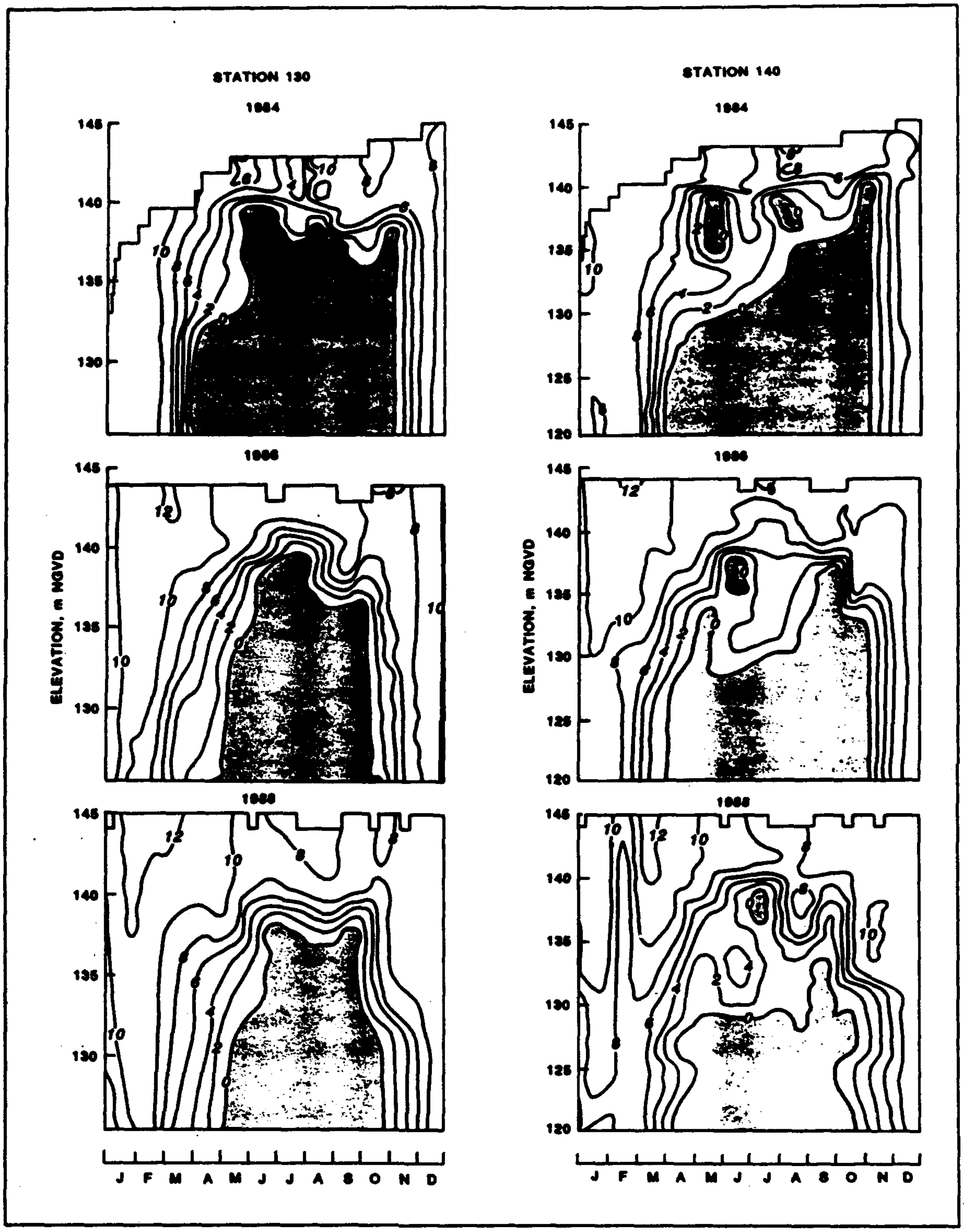

Figure 40. Temporal and vertical patterns in dissolved oxygen (mg/ℓ) for sta 130 and 140, Richard B. Russell Lake, 1984, 1986, and 1988

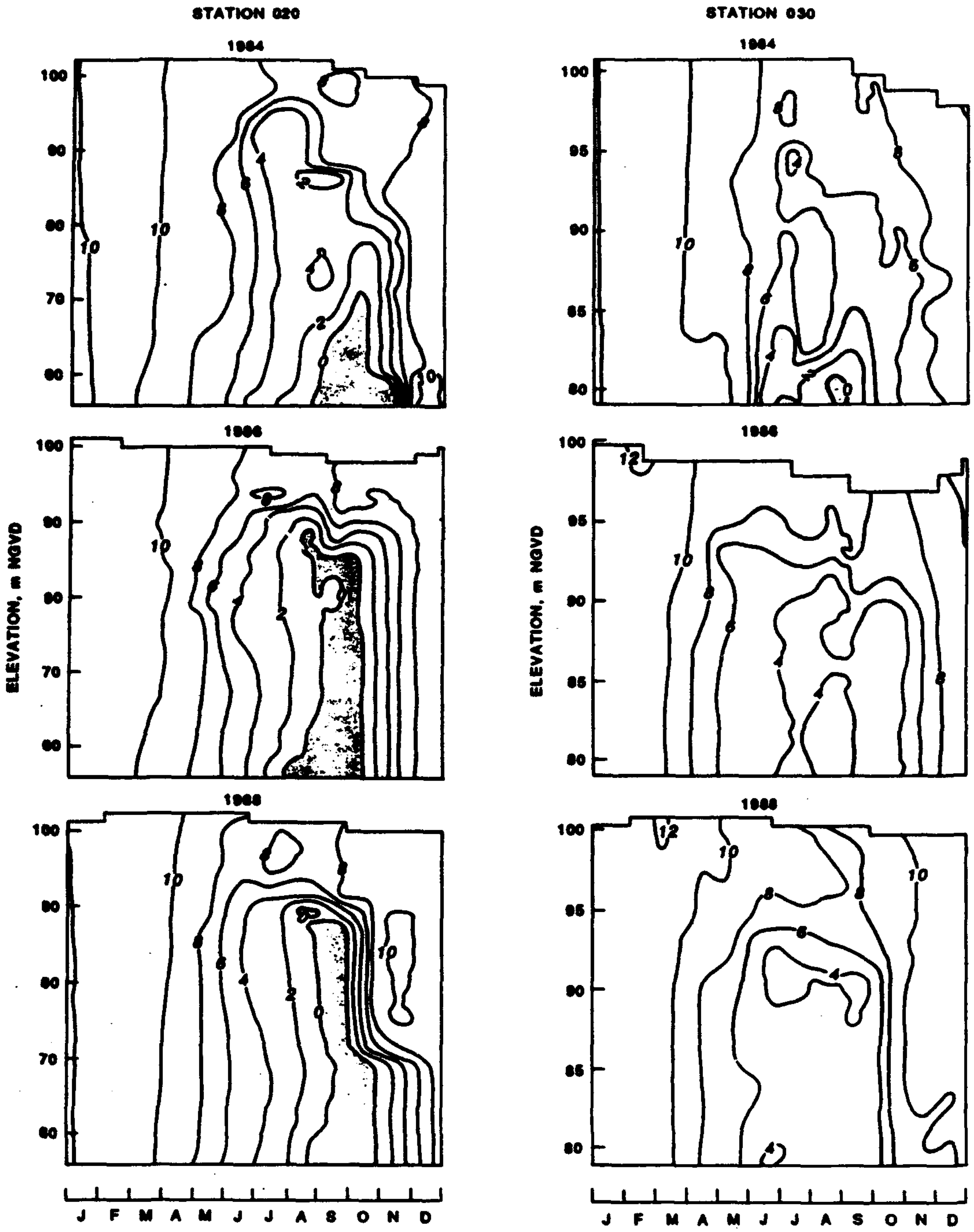

Figure 41. Temporal and vertical patterns in dissolved oxygen (mg/ℓ) for sta 020 and 030, J. Strom Thurmond Lake, 1984, 1986, and 1988

in Richard B. Russell Lake. Minimal annual variation in dissolved oxygen gradients in the Hartwell Lake forebay (Figure 42) suggests downstream variations are primarily influenced by sources other than releases from Hartwell Lake.

The computer program PROFILE was used to calculate volumetric depletion rates at selected stations in Richard B. Russell and J. Strom Thurmond Lakes for 1984-1988 as another means to assess oxygen regimes in the lakes (Table 5). Since selection of dates and depths necessary for calculations is subjective, rigorous annual comparisons of rates are inappropriate. Additionally, volumetric depletion rates in the hypolimnion were greater, in general, than those for the metalimnion, and provided a more accurate estimate of oxygen depletion. Hence, a range of values was calculated for most of the stations and a rate based on a linear regression of appropriate dates (those indicating oxygen depletion and prior to anoxia) is reported. Depletion rates in the main stem of each lake displayed annual variability but were relatively lower than rates in the tributary embayments, suggesting differences in oxygen regimes in each lake are primarily between the main stems and embayments. Explanations of different oxygen depletion rates in the main stems and tributaries may be related to basin morphometry, area of exposed sediment, amount of inundated biodegradable material, and composition of influent material.

Water quality conditions in releases from Hartwell, Richard B. Russell, and J. Strom Thurmond Dams for the 5-year period reflect conditions in each lake and provide an additional viewpoint in evaluating long-term trends in water quality in the three-lake system. Although, limited data for 1984, 1985, and 1988 restrict detailed annual comparisons at this time, seasonal trends in release water concentrations of dissolved oxygen were most obvious.

Temporal trends in dissolved oxygen concentrations in the outflows of each lake reflect among-lake variation in discharge water quality (Figure 43). Concentrations of dissolved oxygen in the releases from Richard B. Russell Dam during thermal stratification each year remained near 6.0 mg/ℓ. Those for the releases from Hartwell and J. Strom Thurmond Dams were consistently lower. Higher dissolved oxygen concentrations in the releases from Richard B. Russell Dam can be attributed to operation of the oxygenation system in Richard B. Russell Lake. Consequently, continued operation of the oxygenation system will result in continued release of well-oxygenated water from Richard B. Russell Dam.

Concentrations of manganese and iron in the releases from Hartwell, Richard B. Russell, and J. Strom Thurmond Dams for the 5-year study period are depicted in Figures 44 and 45, respectively. Maximum observed concentrations of dissolved manganese in the releases from Richard B. Russell Dam remained near 0.4 mg/ℓ during the first 3 years but were somewhat lower in 1987 and 1988 (0.2 to 0.3 mg/ℓ). Additionally, concentrations of dissolved manganese in the releases from Hartwell Dam were lower in 1988 than concentrations observed in 1986 and 1987, and a similar pattern was observed

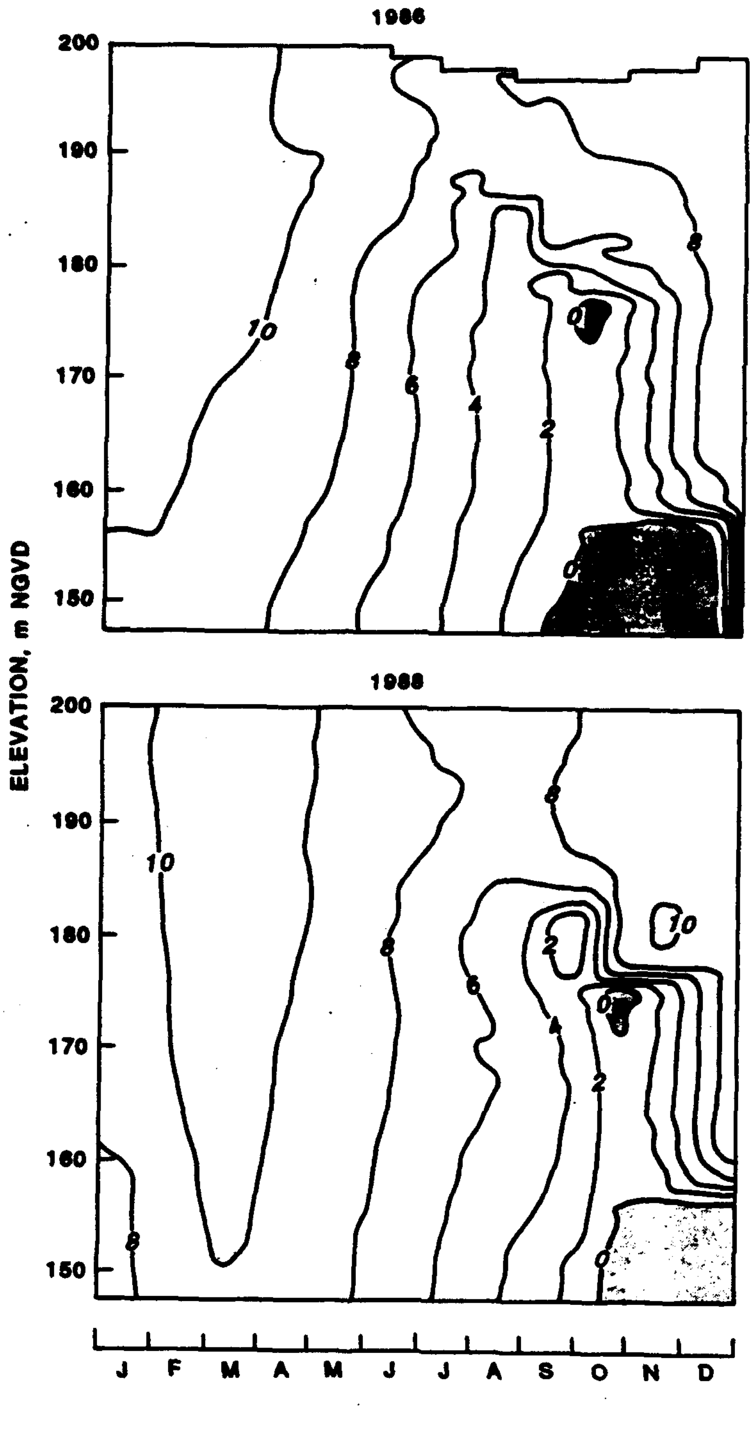

Figure 42. Temporal and vertical patterns in dissolved oxygen (mg/ℓ) for sta 210, Hartwell Lake, 1986 and 1988

Table 5
Volumetric Depletion Rates (mg l^{-1} day^{-1}) Richard B. Russell and J. Strom Thurmond Lakes Based on PROFILE Calculations

Station	1984	1985	1986	1987	1988
020	0.058	0.058	0.053	0.062	0.063
030	0.039	0.031	0.025	0.037	0.062
120	0.033	0.051	0.069	0.064	0.063
130	0.289	0.071	0.058	0.114	0.101
140	0.140	0.108	0.145	0.092	0.105

in the releases from J. Strom Thurmond Dam. While maximum concentrations of particulate iron in the releases from Hartwell Dam were similar for 1984 to those for 1986, increased maximum concentrations were observed in 1987 and 1988. In contrast, maximum concentrations of particulate iron in the releases from Richard B. Russell Dam were lower for 1987 and 1988 than for previous years. In the releases from J. Strom Thurmond Dam, particulate iron concentrations were highest in 1984 and midwinter of 1986-1987. Particulate iron concentrations observed in 1985, 1986, and 1988 were relatively lower. With the exception of concentrations in the releases from Richard B. Russell Dam in 1986 and Hartwell Dam in 1987, dissolved iron concentrations in the release waters were negligible during the study period.

Although concentrations of metals in release waters varied annually and rigorous comparison between years is inappropriate with existing data, general temporal and spatial trends were apparent. Lower manganese and iron concentrations in 1987 and 1988 in the releases from Richard B. Russell Dam suggest a reduction in metals export from the lake. However, rigorous assessment of metals loading and export is not possible with available data.

Dissolved manganese and particulate iron were the dominant forms in release waters, reflecting influences of internal oxidation processes. As previously mentioned, oxidation effects of the oxygenation system on the cycling of iron are more pronounced in Richard B. Russell Lake than on manganese cycling, and system operation impacts the transport and deposition of these metals. While the effects of system operation on the transport of manganese and iron in the three-lake system is not easily quantified, initial assessment of system operation is possible. System operation has varied each year, with the upstream system primarily operated in 1985, the downstream system in 1986, and both systems in 1987 and 1988 (the upstream system was turned on in mid-July in 1987 and in August in 1988). As a result of varied system operation, deposition patterns of oxidized iron vary and influence transport of particulate iron from Richard B. Russell Lake. Specifically, during operation of the downstream system, oxidation of iron occurs closer to the hydraulic withdrawal zone and higher concentrations of particulate iron occur in release waters than during operation of the upstream system. When the upstream system is operated, iron oxidation also occurs but increased precipitation during transport to the hydraulic withdrawal zone results in lower release concentrations and less transport.

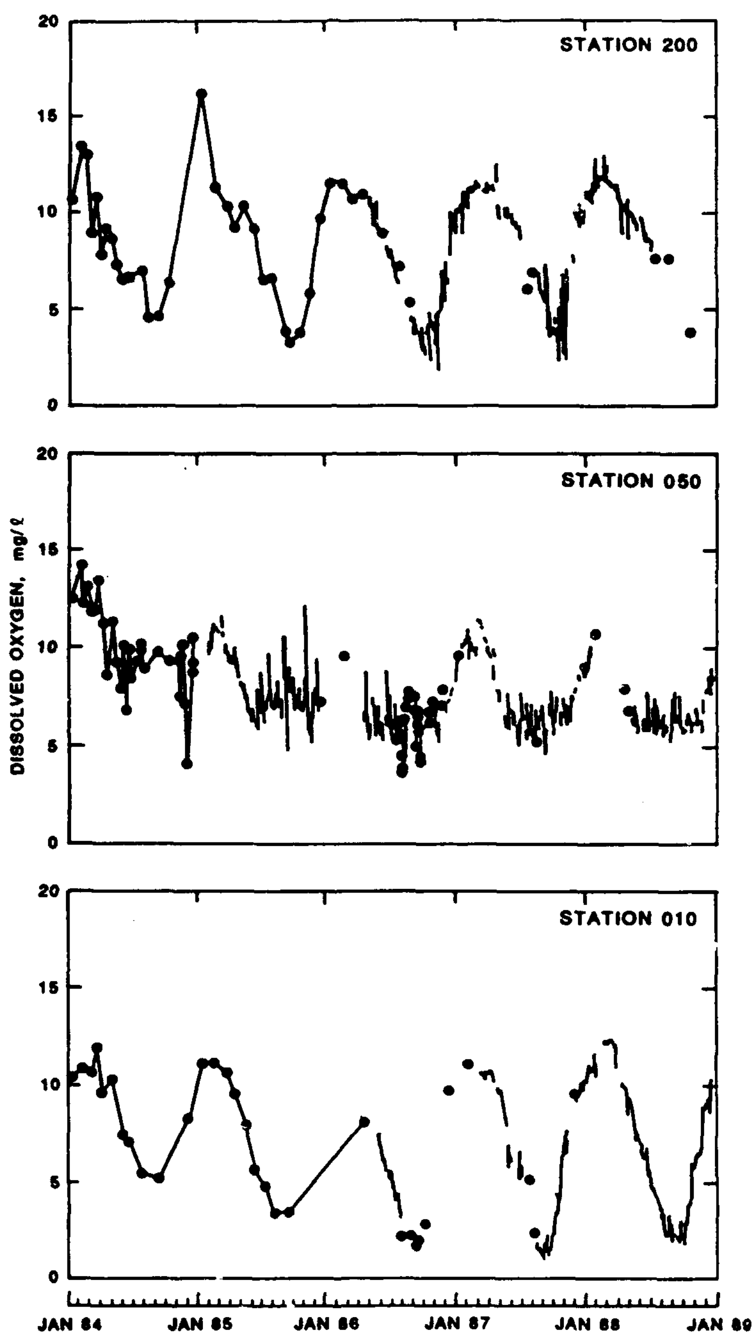

Figure 43. Dissolved oxygen concentrations in the release waters from Hartwell Dam (sta 200), Richard B. Russell Dam (sta 050), and J. Strom Thurmond Dam (sta 010), 1984-1988

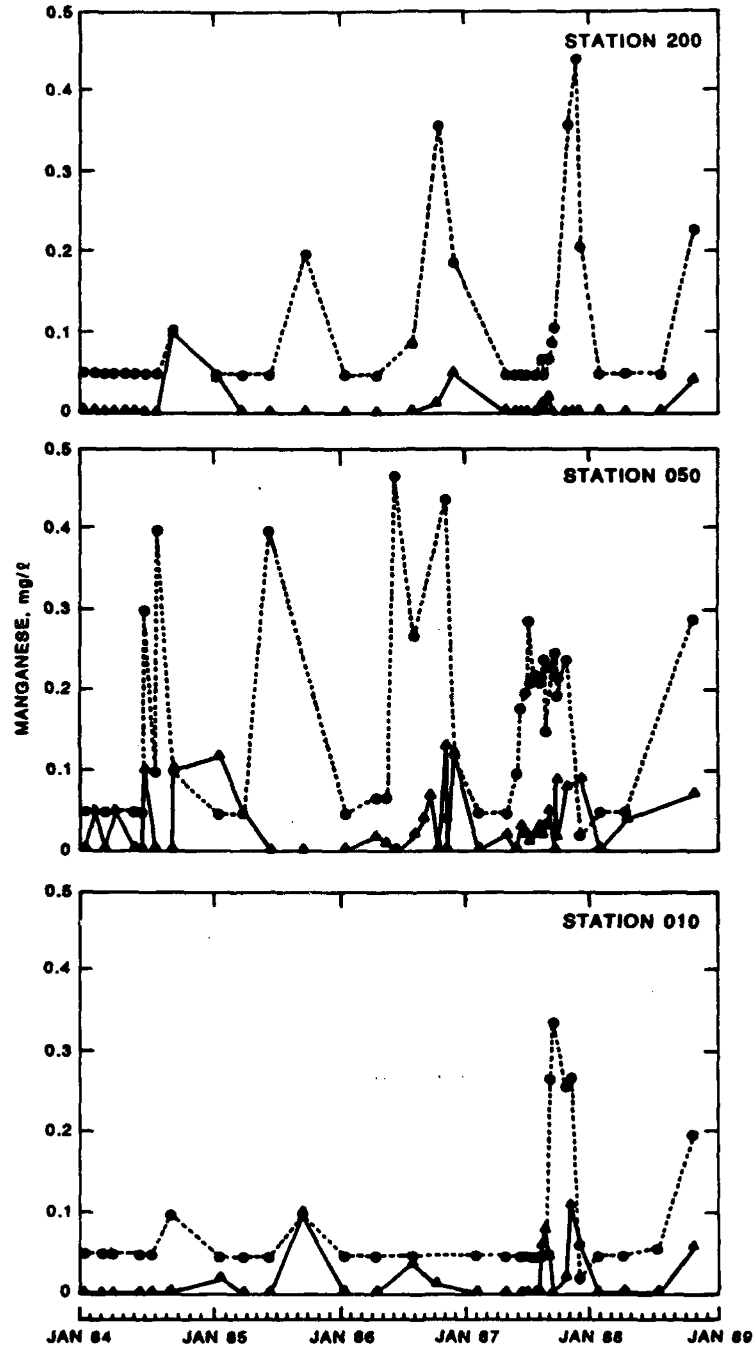

Figure 44. Dissolved and particulate manganese concentrations (mg/ℓ) in the release waters from Hartwell Dam (sta 200), Richard B. Russell Dam (sta 050), and J. Strom Thurmond Dam (sta 010), 1984-1988

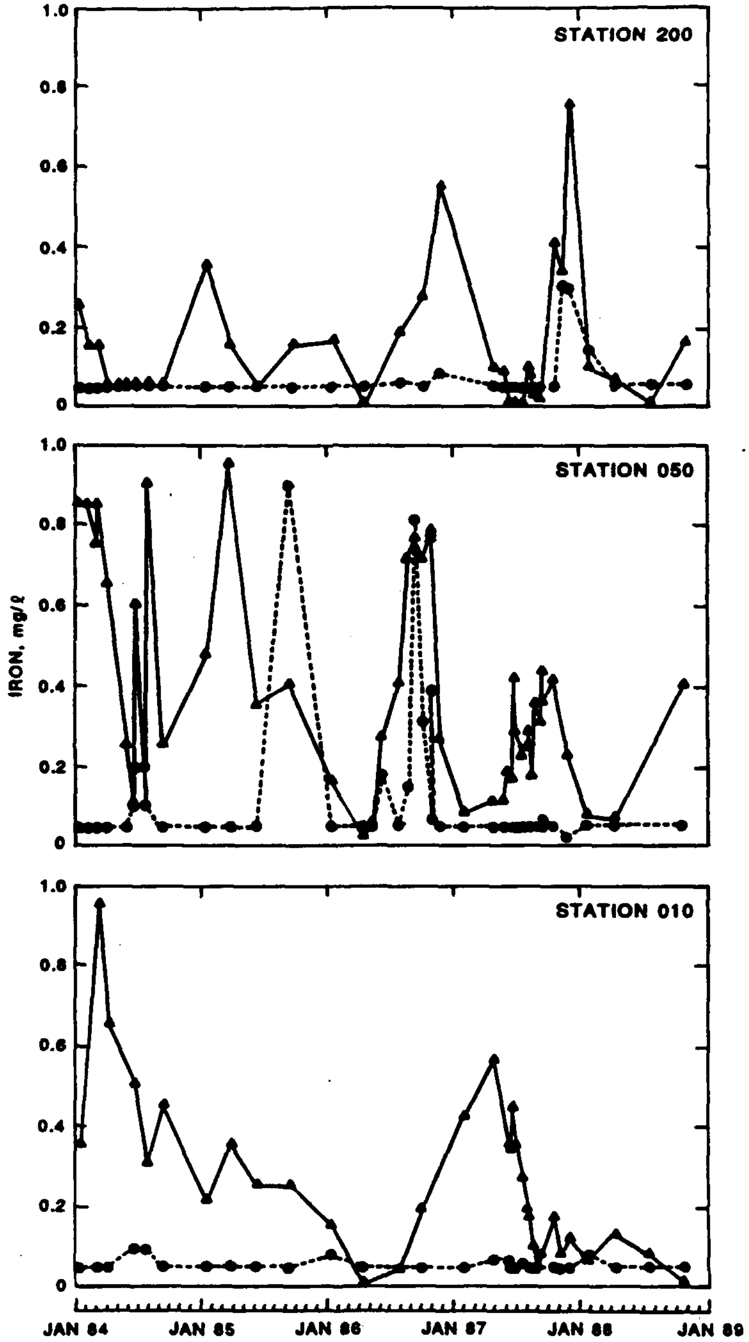

Figure 45. Dissolved and particulate iron concentrations (mg/ℓ) in the release waters from Hartwell Dam (sta 200), Richard B. Russell Dam (sta 050), and J. Strom Thurmond Dam (sta 010), 1984-1988

6 Conclusions

Pronounced changes in water quality in Richard B. Russell Lake following impoundment occurred primarily in the summer of 1984, the first year of impoundment. Temporal trends in water quality observed in Richard B. Russell Lake after the first year of impoundment were similar, however. While concentrations of total organic carbon, phosphorus, and nitrogen were similar in post-impoundment years, spatial differences in concentrations of manganese, iron, and chlorophyll varied somewhat during the study.

Manganese and iron dynamics are the most pronounced of observed chemical parameters. Manganese and iron concentrations increase seasonally in the hypolimnion but respond dissimilarly to oxidation processes. Iron, which is oxidized more readily than manganese, is transported primarily as particulates. Conversely, manganese is transported in the three-lake system primarily in the dissolved form. Insufficient data for calculation of mass balances preclude evaluation of metals transported from the three-lake system. However, Richard B. Russell Lake retains some quantity of particulate iron, due to operation of the oxygenation system, thereby potentially reducing transport downstream.

Concentration peaks of chlorophyll, observed in spring and late summer, suggest temporal variation previously not discernible in the lakes. Although biomass estimates of phytoplankton were not conducted, increased chlorophyll concentrations may be attributed to increased phytoplankton biomass. Identification of phytoplankton species was not conducted either, but seasonal succession of phytoplankton species may have contributed to observed peaks in chlorophyll concentration.

Water quality in the main stem region of Richard B. Russell Lake appears to be improving annually as evidenced by decreasing maximum concentrations of chemical parameters and decreased extent of anoxic regions. Annual variability in the area influenced by the oxygenation system makes assessment of this region difficult, however. Consequently, evaluations of embayment and upstream conditions may be more applicable for assessment of water quality trends. While conditions in the tributary embayments remained similar for the 5-year period, water quality in the upstream region has improved each year since impoundment. Differences in water quality between main stem and embayment stations suggest that inflow quality to these regions has a pronounced effect on water quality within each region. However, discharge

concentrations reflected conditions in the forebay region of the lake, suggesting influences of tributary water quality on discharge water quality are dampened in the main stem.

Operation of the oxygenation system greatly influences water quality above and below Richard B. Russell Dam. Operation of the oxygenation system successfully maintained dissolved oxygen concentrations near the target level of 6 mg/ℓ in the releases of Richard B. Russell Dam throughout the period of stratification. Calculations of dissolved oxygen concentrations in the releases with and without oxygenation indicate that the oxygenation system increases dissolved oxygen concentrations approximately 2 mg/ℓ under current operating conditions. As mentioned previously, the oxidation and resultant precipitation of reduced (i.e. dissolved) iron is enhanced due to operation of the oxygenation system. Effects of the system on reduced or dissolved manganese are less pronounced, however.

Annual trends in the distribution of temperature and dissolved oxygen in the main stem region of J. Strom Thurmond Lake were observed. Hypolimnetic temperatures in J. Strom Thurmond Lake are approximately 2 °C lower since 1984 due to cooler inflows from Richard B. Russell Dam. Dissolved oxygen concentrations in the hypolimnion appear to be lower in the forebay region but are greater in the midstream region since 1984. Possible explanations for these changes include effects of impoundment and subsequent changes in inflow quality, annual variability, and effects of low inflow periods experienced in 1986, 1987, and 1988.

7 Recommendations

Monitoring of physical, chemical, and biological parameters in Hartwell, Richard B. Russell, and J. Strom Thurmond Lakes should be continued to allow further evaluation of impacts of impoundment of Richard B. Russell Lake on the water quality of J. Strom Thurmond Lake. Of particular interest, is the effect of the oxygenation system on release and downstream water quality. Additionally, seasonal in situ monitoring in the downstream region of Richard B. Russell Lake and in release waters should be maintained to continue assistance in operation of the oxygenation system.

Lastly, water quality sampling should continue to address water quality processes in the tailwater and forebay region of Richard B. Russell Dam and Lake to provide a baseline of water quality conditions prior to initiation of pumped-storage operations. Changes in the timing, duration, and velocity of flows resulting from pumped-storage operation may impact nutrient and metal distribution and phytoplankton and zooplankton dynamics in the Richard B. Russell Lake forebay and J. Strom Thurmond Lake headwater region. Pumped-storage operations may provide an upwelling of nutrients to surface waters and affect changes in nutrient and metal distributions. Phytoplankton and zooplankton response to hydraulic changes and varied distribution of nutrients may include changes in production and species composition of the planktonic communities. Since changes in phytoplankton and zooplankton populations may impact fisheries, water quality sampling should continue in the upstream region of J. Strom Thurmond Lake. Finally, the effectiveness of the oxygenation system in the downstream region of Richard B. Russell Lake and headwater region of J. Strom Thurmond Lake should be determined following initiation of pumped-storage operation.

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Appendix A
Analytical Procedures

Quality Control

Replicate samples were obtained from four to six randomly selected sampling locations during each sampling trip. These samples, which generally represented approximately 10 percent of the total number of samples, provided a means for estimating errors due to sampling and intrinsic variability. Coefficients of variation (CV) have been calculated for each variable for each replicate and then a mean of the CV's has been calculated. These means (Table A1) represent the relative sampling precision and provide a method for comparing different analytical procedures.

The analytical precision of each assay was evaluated by splitting samples in the laboratory and analyzing each subsample separately. As with replicates,

Table A1
Mean Coefficients of Variation for Replicate and Split Samples

Variable	Replicate Samples	Split Samples
Total alkalinity	1.9 (25)[1]	3.9 (23)
Total organic carbon	3.2 (24)	1.9 (23)
Dissolved organic carbon	4.9 (24)	1.7 (23)
Total phosphorus	6.6 (24)	7.0 (22)
Total soluble phosphorus	2.9 (24)	3.2 (22)
Soluble reactive phosphorus	3.9 (24)	3.4 (22)
Total nitrogen	9.6 (24)	11.5 (20)
Total dissolved nitrogen	9.9 (23)	9.0 (23)
Ammonia nitrogen	2.1 (24)	3.6 (23)
Nitrate-nitrite nitrogen	1.5 (22)	1.9 (23)
Total iron	9.8 (24)	8.1 (23)
Dissolved iron	17.9 (22)	0.4 (23)
Total manganese	6.3 (24)	1.3 (23)
Dissolved manganese	3.5 (23)	0.9 (23)

[1] Number of samples denoted in parentheses.

split samples were randomly selected for each sampling period. These samples provided a test of analytical reliability and a measure of the normal variability due to analysis. The CV was calculated for each split which had values above the detection limit and mean CV's were calculated for each variable (Table A1).

The accuracy, or description of how closely analyzed values are to the actual values, was evaluated by the analysis of spiked samples prepared in the laboratory. Laboratory values were compared with spike values and recorded as percent recovery (i.e., the lab value expressed as a percent of the actual value). These results are presented in Table A2.

Table A2
Mean Percent Recovery for Laboratory Spiked Samples

Variable	Mean Percent Recovery
Total alkalinity	94.2 (24)[1]
Total organic carbon	101.1 (24)
Total phosphorus	93.2 (24)
Total soluble phosphorus	130.3 (23)
Soluble reactive phosphorus	96.7 (24)
Total nitrogen	106.7 (24)
Total dissolved nitrogen	109.8 (23)
Ammonia nitrogen	92.7 (23)
Nitrate-nitrite nitrogen	98.1 (24)
Total iron	98.0 (24)
Dissolved iron	84.5 (24)
Total manganese	107.5 (18)
Dissolved manganese	102.1 (18)

[1] Number of samples denoted in parentheses.

Analytical Methods

The analytical methods, detection limits, equipment used, and calibration methods are described below.

Water column depth

Method: Depth sounding.
Detection Limit: 0.1 m.

Secchi disc transparency

Method: Mean depth of disappearance and reappearance of disc.
Detection Limit: 0.1 m.
Equipment: 20-cm Secchi disc with alternating black and white quadrants.

Water temperature[1]

Method: Thermistor thermometer.
Detection Limit: 0.1 °C.
Calibration: National Bureau of Standards certified thermometer.

Dissolved oxygen[1]

Method: Membrane electrode.
Detection Limit: 0.1 mg/ℓ.
Calibration: Air calibration.
Reference: Hydrolab Corp. (1985)[2].

Specific conductance[1]

Method: Electrometric.
Detection Limit: 1 µS.
Calibration: Conductivity standard solutions.
Reference: Hydrolab Corp. (1985).
Comments: All readings were corrected for temperature to 25 °C.

pH[1]

Method: Electrometric.
Detection Limit: 0.1 pH unit.
Calibration: Determination of pH with pH 7 and pH 4 buffer solutions.
Reference: Hydrolab Corp. (1985).

Oxidation-reduction potential[1]

Method: Electrometric.
Calibration: Ferric/ferrous iron solution standardized to 475 mV.
Reference: Hydrolab Corp. (1985).

Alkalinity

Method: Potentiometric titration.
Detection Limit: 1.0 mg/ℓ as CaCO3.
Calibration: pH meter; Beckman Model Zeromatic IV (Beckman Instruments Inc., Fullerton, CA).

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1 *In situ* measurements made with Hydrolab Surveyor.

2 References cited in this appendix are located at the end of the main text.

Reference: American Public Health Association (1980).
Sample Handling: Analyzed within 24 hr of sample collection.

Carbon

A. Total Organic Carbon (TOC)
Method: Acid-sparge; infrared analysis.

B. Total Filterable Organic Carbon (DOC)
Method: Acid-sparge; infrared analysis on sample filtered through a glass fiber filter.

Detection Limits: 0.2 mg C/ℓ.
Calibration: Per manufacturer's guidelines; standard curves.
Equipment: Carbon analyzer (Beckman Model 915B, Beckman Instruments Inc., Fullerton, CA).
Reference: U.S. Environmental Protection Agency (1979).
Sample Handling: Stored at 4 °C prior to analyses. Filtered on day of collection. Analyses performed within 2 weeks.

Phosphorus

A. Total Phosphorus (TP)
Method: Sulfuric acid-persulfate oxidation digestion; automated ascorbic acid calorimetric method.

B. Total Soluble Phosphorus (TSP)
Method: Sulfuric acid-persulfate oxidation digestion on sample filtered through 0.45-micron membrane filter; automated ascorbic acid calorimetric method.

C. Soluble Reactive Phosphorus (SRP)
Method: Automated ascorbic acid calorimetric methods after filtration through a 0.45-micron membrane filter.

Detection Limits: 0.005 mg P/ℓ (dependent upon range used in analyses).
Calibration: Standard curves at beginning and end of each batch of samples.
Equipment: Autoanalyzers (Technicon Auto Analyzer II, Technicon Instruments Corp., Tarrytown, NY).
Reference: American Public Health Association (1980).
Sample Handling: Stored at 4 °C prior to analysis, filtered day of collection. Anoxic samples filtered in field and held anoxic in syringes. Digestion on day of collection. SRP analyzed within 48 hr of collection. TP and TSP analyzed within 72 hr of collection.

Nitrogen

A. Total Nitrogen (TN)
   Method: Sulfuric acid persulfate oxidation digestion; DeVarda's Alloy reduction (Raveh and Avnimelech 1979); automated phenol-hypochlorite calorimetric method.

B. Total Soluble Nitrogen (TSN)
   Method: Same as above except sample was filtered through a 0.45-micron membrane filter prior to digestion.

C. Ammonia Nitrogen (NH_4-N)
   Method: Automated phenol-hypochlorite calorimetric method.

D. Nitrate-Nitrite Nitrogen (NO_3/NO_2-N)
   Method: Automated cadmium reduction calorimetric method, sample filtered through a 0.45-micron membrane filter prior to analysis.

Detection Limits: 0.02 mg N/ℓ for TN, TSN, and NH_4-N; 0.04 mg N/ℓ for NO_3/NO_2-N (dependent upon range used in analysis).
Calibration: Standard curves at beginning and end of each batch of samples.
Equipment: Autoanalyzers (Technicon Auto Analyzer II, Technicon Instrument Corp., Tarrytown, New York).
Reference: American Public Health Association (1980).
Sample Handling: Stored at 4 °C prior to analyses, filtered day of collection. Anoxic samples filtered in the field and held anoxic in syringes. Digestion on day of collection. DeVarda's alloy added 16 to 20 hr prior to analyses. NH_4-N and NO_3/NO_2-N analyzed within 48 hr of sample collection. TN and TSN analyzed within 96 hr of sample collection.

Comments: TN and TSN analyses performed on samples digested for TP and TSP, respectively (i.e. one digestion for both elements).

Sulfide

Method: Lead sulfide.
Detection Limit: 0.1 mg/ℓ.
Reference: Hach Chemical Co. (1978).

Metals

A. Total Iron and Manganese (TFe, TMn)
   Method: Hydrochloric/nitric acid reflux digestion, atomic absorption spectrophotometry.

B. Dissolved Iron and Manganese (DFe, DMn)

Method: Sample filtered through a 0.1-micron membrane filter, atomic absorption spectrophotometry.

Detection Limits: 0.05 mg/ℓ.
Calibration: Standard curves per manufacturer's guidelines.
Equipment: Atomic absorption spectrophotometer (Model 4000, Bodenseewerk Perkin-Elmer, and Company, Uberlingen, West Germany).
Reference: American Public Health Association (1980).
Sample Handling: Filtered, acidified (pH<2), and stored at 4 °C. Anoxic samples filtered in field and held anoxic in syringes. Digestion and analysis within 72 hr of collection.

Chlorophyll *a*

As in previous years of this study, samples for analysis of chlorophyll *a* were taken as integrated depth samples at depths equivalent to twice the Secchi depth. Chlorophyll *a* and other phytopigments were extracted using a modified procedure which employs dimethyl-formamide as the solvent. This method is more rapid and efficient and has been previously described (Speziale et al. 1984; Hains 1985). Otherwise the analysis is identical to the procedure used in previous years of this study.

REPORT DOCUMENTATION PAGE

Form Approved
OMB No. 0704-0188

1. AGENCY USE ONLY *(Leave blank)*

2. REPORT DATE
June 1994

3. REPORT TYPE AND DATES COVERED
Final report

4. TITLE AND SUBTITLE
Water Quality Studies: Richard B. Russell and J. Strom Thurmond Lakes; Summary Report

5. FUNDING NUMBERS
Intra-Army Order
No. PD-EI-84-07

6. AUTHOR(S)
Steven L. Ashby, Robert H. Kennedy, Joe H. Carroll
John J. Hains

7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES)
U.S. Army Engineer Waterways Experiment Station
3909 Halls Ferry Road, Vicksburg, MS 39180-6199

8. PERFORMING ORGANIZATION REPORT NUMBER
Miscellaneous Paper
EL-94-6

9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESS(ES)
U.S. Army Engineer District, Savannah, P.O. Box 889
Savannah, GA 31402-0889

10. SPONSORING/MONITORING AGENCY REPORT NUMBER

11. SUPPLEMENTARY NOTES
Available from National Technical Information Service, 5285 Port Royal Road, Springfield, VA 22161

12a. DISTRIBUTION/AVAILABILITY STATEMENT
Approved for public release; distribution is unlimited.

12b. DISTRIBUTION CODE

13. ABSTRACT *(Maximum 200 words)*

This report is the fifth in a series of annual interim reports documenting the results of a comprehensive water quality study at Hartwell and Richard B. Russell Lakes and at J. Strom Thurmond Dam and Reservoir (formerly called Clarks Hill Dam and Lake) in the Savannah River Bas and pertains to the period January to December 1988. This report also summarizes the results of all studies during the 5-year period following impoundment of Richard B. Russell Lake.

Water quality conditions in Richard B. Russell Lake were markedly improved in 1988 compared with those of early study years. Most pronounced was a marked decline in the extent of the anoxic region in the main stem of the lake. Coincidentally, concentrations of manganese and iron in the main stem of the lake and in release waters were lower than concentrations observed in previous years. Conditions in the major tributary embayments were similar to those of previous years, however. Continued improvement in water quality is anticipated over the next several years, due to continued reduction in the decomposition of organic material inundated during impoundment.

Dissolved oxygen concentrations in Richard B. Russell Lake releases during summer months were maintained near 6 mg/ℓ with operation of the oxygenation system.

(Continued)

14. SUBJECT TERMS

Hartwell Lake	Oxygenation	Savannah River
J. Strom Thurmond Lake	Reservoir	Water quality
Limnology	Richard B. Russell Lake	

15. NUMBER OF PAGES
82

16. PRICE CODE

17. SECURITY CLASSIFICATION OF REPORT	18. SECURITY CLASSIFICATION OF THIS PAGE	19. SECURITY CLASSIFICATION OF ABSTRACT	20. LIMITATION OF ABSTRACT
UNCLASSIFIED	UNCLASSIFIED		

13. (Concluded).

Water quality in J. Strom Thurmond Lake was similar to previous study years; however, spatial trends in temperature and dissolved oxygen concentrations were observed. Cooler hypolimnetic temperatures and a more pronounced thermocline suggest that the thermal structure in the lake is more defined than that observed prior to impoundment of Richard B. Russell Lake. Hypolimnetic dissolved oxygen concentrations have increased in the mid- and upstream regions of J. Strom Thurmond Lake, but concentrations in the hypolimnion of the downstream region of the lake have declined since 1984.

Manganese and iron continue to be the most pronounced of chemical parameters currently monitored. Transport of dissolved manganese and particulate iron in the three-lake system is apparent, but the implications of effects on water quality are less discernible.